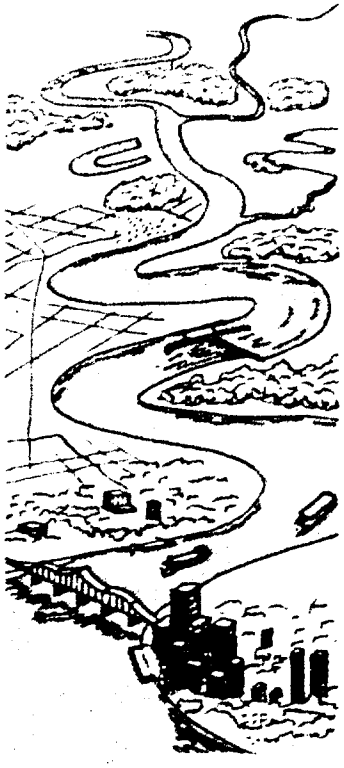
US Army Corps of Engineers



ENVIRONMENTAL AND WATER QUALITY OPERATIONAL STUDIES

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SUPPLEMENTAL LIMNOLOGICAL STUDIES AT RICHARD B. RUSSELL AND CLARKS HILL LAKES, 1983-1985

by

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19 ABSTRACT (Continue on reverse if necessary and identify by block number)

A 3-year, comprehensive water quality study was initiated in 1983 at Richard B. Russell and Clarks Hill Lakes, Georgia-South Carolina. Objectives of the study were to evaluate water quality changes in both lakes following the impoundment of Richard B. Russell Lake and to monitor the performance of an oxygenation system installed in the forebay of Richard B. Russell Lake. In addition to this study, supplemental studies were conducted to provide more detailed information concerning processes influencing water quality conditions in the two lakes. Results of these studies, which centered on establishment of biological communities, decomposition, nutrient and metal dynamics, sediment/water exchanges, and interactions between the lakes, were presented at a symposium held 5-6 February 1985 at Hickory Knob State Park, McCormick, S. C. This report documents results presented at the symposium.

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PREFACE

A 3-year, comprehensive water quality study at Richard B. Russell and Clarks Hill Lakes was initiated in 1983 as a joint effort by the US Army Engineer District, Savannah, and the US Army Engineer Waterways Experiment Station (WES). The study was funded through Intra-Army Order No. PD-EI-84-07. The purpose of the study was to document postimpoundment water quality and environmental conditions in both lakes and to evaluate the performance of an oxygen-injection system installed in Richard B. Russell Lake. This report describes supplemental research activities conducted at both sites. These activities were designed to allow more complete description of processes and events influencing reservoir water quality.

These latter studies were sponsored by the Office, Chief of Engineers (OCE), US Army, through the Environmental and Water Quality Operational Studies (EWQOS) Program and the Water Operations Technical Support (WOTS) Program. The OCE Technical Monitors for EWQOS were Dr. John Bushman, Mr. Earl Eiker, and Mr. James Gottesman. Dr. J. L. Mahloch, WES, was Program Manager for the EWQOS and WOTS Programs.

This report was prepared under the direct supervision of Dr. Robert H. Kennedy, Aquatic Processes and Effects Group (APEG), Ecosystem Research and Simulation Division (ERSD), Environmental Laboratory (EL), WES. Dr. Thomas L. Hart was Chief, APEG; Mr. Donald L. Robey was Chief, ERSD; and Dr. John Harrison was Chief, EL. The report was edited by Ms. Jessica S. Ruff of the WES Information Products Division.

Commander and Director of WES is COL Dwayne G. Lee, CE. Dr. Robert W. Whalin is Technical Director.

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CONTENTS

CONVERSION FACTORS, NON-SI TO SI (METRIC)
UNITS OF MEASUREMENT

Non-SI units of measurement used in this report can be converted to SI (metric) units as follows:

Multiply	By	To Obtain
acres	4,046.873	square metres
feet	0.3048	metres
inches	2.54	centimetres
ounces (US fluid)	0.02957353	cubic decimetres
pounds (force) per square inch	6.894757	kilopascals
square feet	0.09290304	square metres
tons (2,000 pounds, mass)	907.1847	kilograms
tons (mass) per acre	0.22	kilograms per square metre

SUPPLEMENTAL LIMNOLOGICAL STUDIES AT RICHARD B. RUSSELL AND CLARKS HILL LAKES, 1983-1985

PART I: INTRODUCTION*

Background

1. Richard B. Russell Dam, which was authorized as Trotters Shoals Dam by the "Flood Control Act of 1966," Public Law 89-789, Eighty-Ninth Congress, HR 18233, was placed into operation in late 1983 when final closure of the gates led to the impoundment of Richard B. Russell lake. The project, which is located on the Savannah River near Calhoun Falls, S. C. (Figure I-1), was designed to provide power generation, incidental flood control, recreation, streamflow regulation, and water supply. The lake covers approximately 26,000 acres* of previously forested land, stretching from the tailwaters of Hartwell Dam to the headwaters of Clarks Hill Lake. In addition to the Savannah River, two other major tributaries are impounded: Beaverdam Creek and Rocky River.

2. Richard B. Russell Dam is a concrete structure providing for near-surface releases through tainter gates and middepth releases for hydroelectric power generation. The powerhouse contains four conventional turbines, which are currently in operation, and four pumped-storage turbines, which will be operational by the early 1990's. Richard B. Russell Lake is large and morphologically complex, with mean and maximum depths of 12 and 47 m, respectively.

3. A number of environmental and water quality questions were raised prior to completion of the project. Most prominent were questions concerning the immediate effects on water quality of the inundation and decay of terrestrial material, and the impacts of Russell

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* Part I was written by Robert H. Kennedy, Environmental Laboratory, US Army Engineer Waterways Experiment Station, Vicksburg, Miss.

** A table of factors for converting non-SI units of measurement to SI (metric) units is presented on page 3.

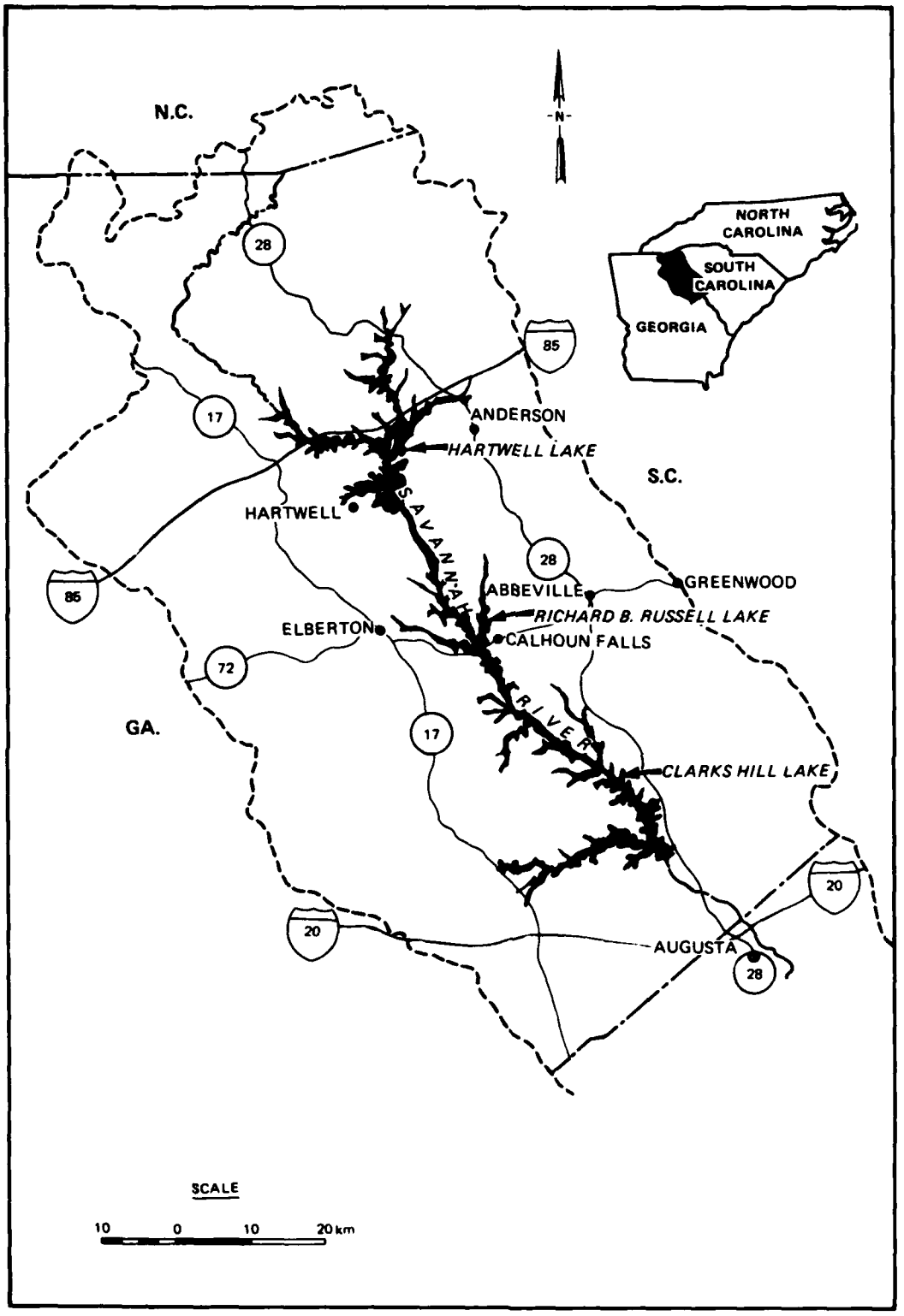

Figure I-1. Project location and vicinity map

Lake on Clarks Hill Lake. These concerns prompted the US Army Engineer District, Savannah (SAS), to initiate several efforts to identify and/or solve potential water quality problems prior to filling of the lake. These included:

a. Formation of a technical committee to outline potential problems at the project site.

b. Simulation of the Hartwell-Russell-Clarks Hill Lake system using physical and mathematical models (Smith et al. 1981).

c. Development and testing of an oxygen-injection system capable of replacing oxygen lost from Russell Lake bottom waters so as to ensure that release concentrations would not fall below 6 mg/ℓ.

d. Laboratory evaluation of the oxygen demand and water quality impacts of inundated soils, vegetation, and organic detritus (Gunnison et al. 1984).

e. Studies of preimpoundment conditions in Russell Lake (US Army Corps of Engineers 1981) and Clarks Hill Lake (US Army Corps of Engineers 1982).

4. Cooperative efforts were also initiated by the SAS and the US Army Engineer Waterways Experiment Station to study the water quality of Russell and Clarks Hill Lakes during the first 3 years following impoundment of Russell Lake. The objectives of these studies were to:

a. Evaluate water quality conditions in Russell Lake during and following impoundment.

b. Monitor the performance of an oxygen-injection system installed on the bottom of the forebay area of Russell Lake.

c. Document any water quality impacts to Clarks Hill Lake due to the impoundment of Russell Lake.

Results of the first year of the study are documented in James et al. (1985). The locations of sampling stations for this study are presented in Figures I-2 and I-3.

5. Because of a general interest in better defining processes of importance in determining water quality in Corps of Engineer reservoirs, additional studies were initiated at the site under the sponsorship of the Environmental and Water Quality Operational Studies Program (Keeley et al. 1978) and the Water Operations Technical Support Program. The

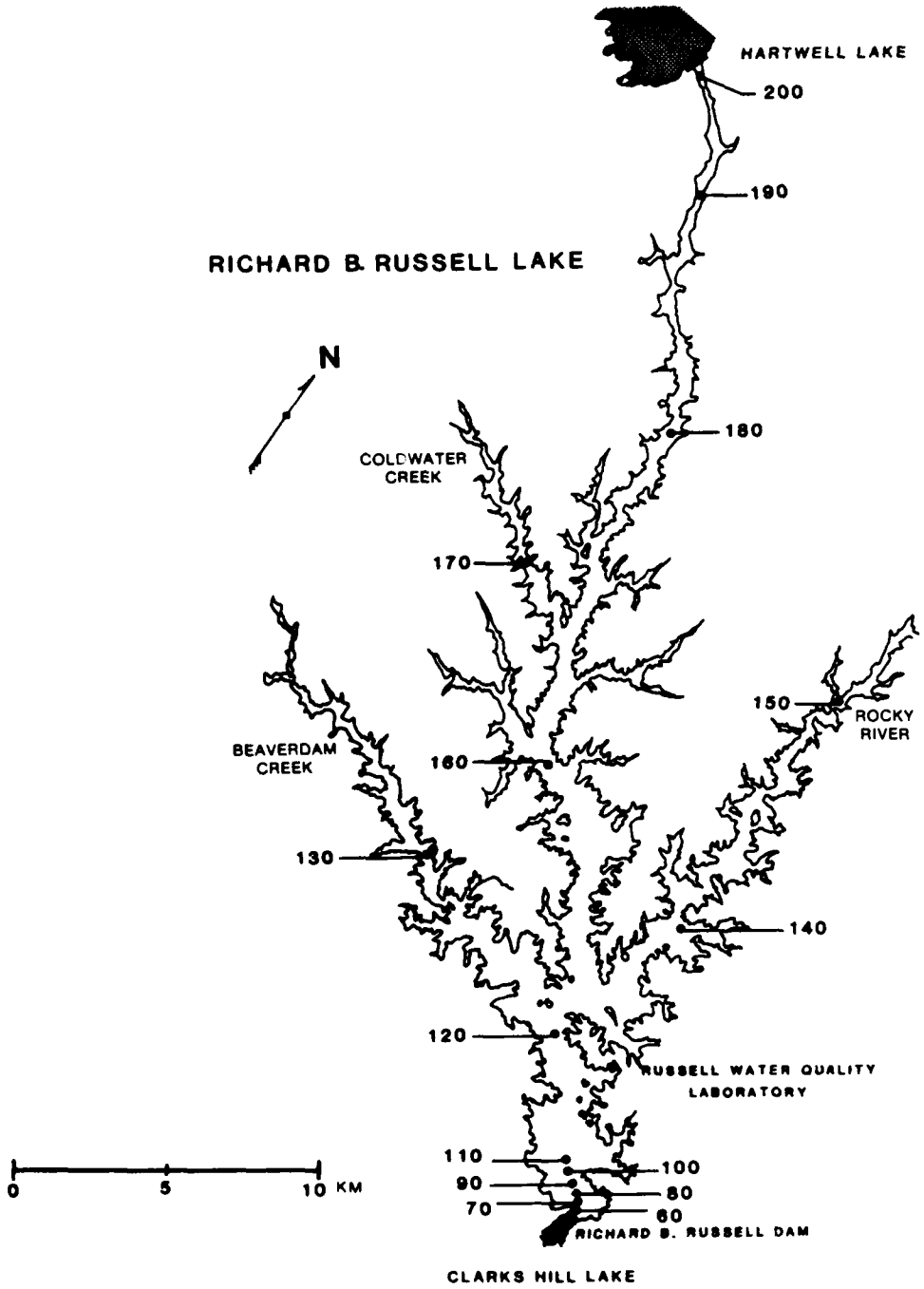

Figure I-2. Location of water quality sampling stations on Richard B. Russell Lake. Numbers identify sampling stations

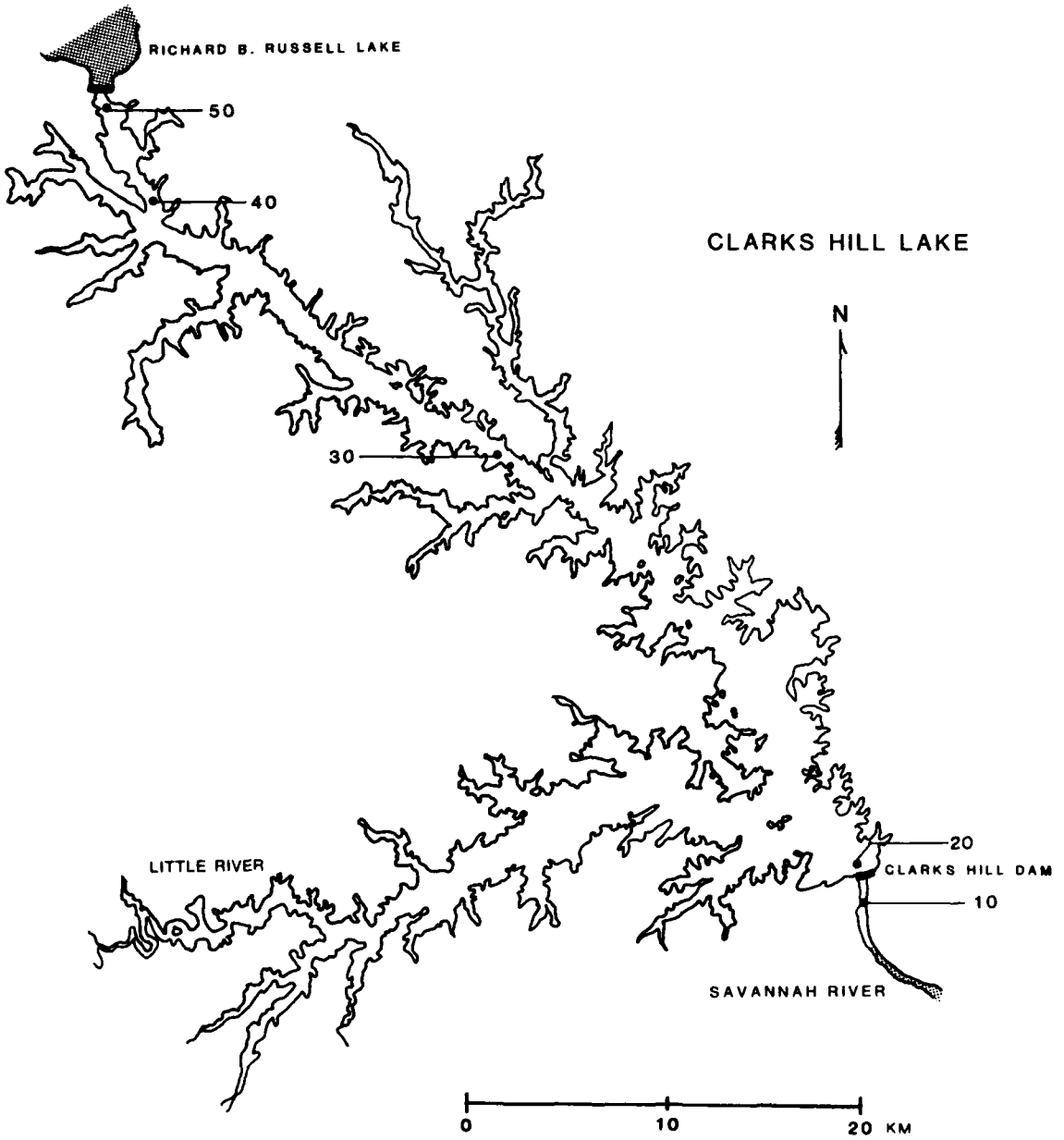

Figure I-3. Location of water quality sampling stations on Clarks Hill Lake. Numbers identify sampling stations

purpose of these studies was to provide additional information concerning specific water quality processes as they affect water quality conditions in Russell Lake and other newly filled reservoirs. Results of these studies, which centered on the establishment of biological communities, decomposition, nutrient and metal dynamics, sediment-water exchanges, and the interactions between lakes, were first presented at a symposium that was held 5-6 February 1985 at the Hickory Knob State Park, McCormick, S. C. The purpose of this report is to provide written documentation of the results presented at that symposium. The reader is directed to the report by James et al. (1985) for detailed information on the study site and the results of monitoring studies at both lakes.

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PART II: NUTRITIONAL SIGNIFICANCE OF DISSOLVED ORGANIC PHOSPHORUS COMPOUNDS IN MEETING PLANKTONIC PHOSPHATE DEMAND IN RICHARD B. RUSSELL RESERVOIR AND CLARKS HILL LAKE*

Introduction

6. Determining nutrient availability is important in understanding those factors that control planktonic community metabolism, for both practical and scientific purposes. The availability of phosphorus most often limits the growth and activities of freshwater communities (Schindler 1977). Dillon and Rigler (1974) showed that in a wide variety of lakes, the annual production of planktonic chlorophyll was correlated to the total phosphorus content of the water. In a more direct manner, both in situ and laboratory bioassays often indicate that addition of P in an available form rapidly stimulates the growth of planktonic organisms (Schindler 1974, Schelske 1984).

7. The purpose of this study was to determine whether phosphorus availability is in biochemically limiting supply in Richard B. Russell Lake (RBR) and Clarks Hill Lake (CHL). Also, because RBR was recently created by inundating substantial forested areas without prior clear-cutting and debris removal, the possibility of significant inputs of dissolved organic phosphorus (DOP) inputs was examined. The effect of these DOP inputs in satisfying phytoplankton phosphate demand in RBR was studied. Also studied was the effect of RBR effluents on the P-dynamics of the planktonic community in CHL.

Materials and Methods

Collection of samples

8. Integrated surface (i.e., above the thermocline) samples were collected on 19 July and 8 August 1984 from main stem sites (Stations 190, 180, 160, 120, and 60) in RBR and from main stem sites

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* Part II was written by Robert T. Heath, Department of Biological Sciences, Kent State University, Kent, Ohio.

(Stations 50 and 40) in CHL (see Figures I-2 and I-3, respectively). On 8 August 1984 samples were also collected from main stem sites (Stations 30 and 20) in CHL. A portion of each sample was filtered through a 0.45-μm Milipore HA filter. Samples of whole and filtered water were placed in polyethylene bottles and shipped in insulated, refrigerated cases to Kent State University via express air freight where they were received within 24 hr of collection.

Limnological variables

9. Temperature (degrees Celsius) and oxygen concentration (milligrams O_2 ℓ^{-1}) were determined electronically at the time of collection. The pH of the water was determined potentiometrically in the laboratory, both before and after shipment, using pH meters calibrated by the two-point method employing standard pH 4 and pH 7 buffers. The conductivity was measured electronically at Kent State University and expressed in micromhos. Turbidity of the whole-water sample was expressed as the absorbance at 550 nm in 1-cm cuvettes.

10. Chlorophyll a (chl a) content of the unfiltered sample was determined according to Standard Methods (American Public Health Association 1975). "Active" chlorophyll was determined by subtracting the calculated quantity of pheophytin from total chlorophyll determined by the two-wavelength (665 and 750 nm) procedure.

11. Fluorescence of the water was determined by placing filtered water into 1-cm cuvettes and irradiating the sample with light having a wavelength of 410 nm. Fluorescence at 530 nm was read in arbitrary fluorescence units using a Turner Model 111 Fluorometer.

Phosphorus content of the water

12. Total phosphorus content of unfiltered (total phosphorus, TP) and filtered (total soluble phosphorus, TSP) water samples were determined by first hydrolyzing all P-compounds to phosphate and then detecting phosphate colorimetrically. Hydrolysis was done by adding 0.5 ml of 10-percent persulfate and 0.1 ml of 10 mol sulfuric acid to 2.5-ml samples. These were incubated for 60 min at 126° C and 2 atm in a pressure cooker. After cooling to room temperature, one or two drops of 1-percent phenolphthalein (weight/volume absolute ethanol) were added,

and the solution was titrated to endpoint with 10 mol NaOH and back titrated to endpoint with 1 mol H_2SO_4. Phosphate concentration was determined colorimetrically in 2.5-ml aliquots of this hydrolysate placed into clean tubes to which were added 0.5 ml molybdate reagent and 0.2 ml ascorbate reagent (US Environmental Protection Agency 1971). Absorbance at 885 nm in 1-cm curvettes was read in a Bausch and Lomb Spectronic 710 digital spectrophotometer. The absorbance of the sample was taken as the mean of triplicate aliquots run in tandem. Endogenous color of the water sample was determined by reading the absorbance at 885 nm of an aliquot run in tandem to which molybdate reagent was added but water was substituted for ascorbate reagent; endogenous color was subtracted from the mean of triplicates to yield the mean color-corrected absorbance of the reaction. The concentration of phosphorus in the samples was determined by comparing the mean color-corrected absorbance with the absorbance of a set of standards of KH_2PO_4 (500, 250, 125, 62.5 µg P ℓ^{-1}) run in duplicate in tandem with the samples.

13. Soluble reactive phosphorus (SRP) was determined as above but omitting prior hydrolysis. That is, to 2.5 ml of filtered water was added 0.5 ml molybdate and 0.2 ml ascorbate reagents. Following 10-min color development, the absorbance at 885 nm was read in a 1-cm cuvette. Endogenous color was determined as above. The mean color-corrected absorbance of triplicate aliquots was used to determine the SRP-P concentration as above. Concentrations were expressed as µg P ℓ^{-1} and converted to nanomolar concentrations (nmol P ℓ^{-1}).

Phosphomonoester
(PME) content of the water

14. The PME concentration was determined as the increase in SRP following incubation of filtered water with alkaline phosphatase. This enzyme specifically hydrolyzes PME to release phosphate detectable as SRP. To 2.25 ml of filtered lake water was added 0.25 ml of a solution containing 0.5 mg calf intestinal mucosa alkaline phosphatase (Sigma) per millilitre 0.1 mol tris-hydroxyomethylaminomethane at pH 9.0. Samples were run in two batches of triplicate portions of filtered water. The SRP of one batch was determined immediately. The other

batch was incubated at 37° C. After 24 hr the SRP was determined on this second batch of triplicates. The concentration of SRP was determined by comparison with a set of standards of KH_2PO_4 run in tandem. To determine that the enzymatic reaction was complete, a set of standards of glucose-6-phosphate (500, 250, 125, 62.5 µg P ℓ^{-1}) was run in tandem.

Phosphatase activity

15. Because the pH of the water was consistently greater than 7, only alkaline phosphatase activity was determined.

16. Phosphatase activity generally exhibits Michaelis-Menten kinetics (Fernley 1971); therefore, the velocity of release of phosphate from PME was determined from the maximal velocity of hydrolysis V_{max}, the K_m, and the PME concentration. The maximal velocity of hydrolysis was determined using p-nitrophenyl phosphate (pNPP) as a model substrate. To 2.5 ml whole lake water was added 0.3 ml of a solution of 3 mg pNPP (Sigma) per millilitre of water, and 0.3 ml of a buffer being 0.1 mol tris-hydroxymethylaminomethane and 0.01 mol $MgCl_2$, pH 9.0. This mixture was incubated at ambient temperature. Activity was determined as the hourly change in absorbance at 395 nm in 1-cm cuvettes. Change in absorbance per hour was divided by 8.932×10^{-6} to express hydrolytic rate in units of nanomoles per litre per hour (nmol ℓ^{-1} hr^{-1}). The value of K_m was determined by the method of Lineweaver and Burk (1934), using concentrations of pNPP ranging from 2.5×10^{-5} mol to 2×10^{-3} mol incubated at 37° C. The velocity of release of phosphate from naturally occurring PME was estimated from solution of the equation

$$v_{rel} = \frac{V_{max}(PME)}{K_m + (PME)}$$

Where V_{max} was expressed in units of nmol ℓ^{-1} hr^{-1} and PME and K_m were in units of nmol ℓ^{-1}, the velocity of release was given in units of nmol ℓ^{-1} hr^{-1}.

Uptake of phosphate by seston

17. The rate of phosphate sorbtion to seston was determined radiometrically by collecting particles on Millipore HA filters that

were pretreated and postrinsed to remove nonparticulate radiolabeled phosphate. Millipore HA filters were presoaked in 0.5 mol KH_2PO_4. Unfiltered 50-ml portions of lake water were incubated in a water bath adjusted to ambient lake temperature. At time zero, 0.05 to 0.2 ml of solution containing 1 to 2 μ Ci (i.e., about 1×10^3 Bq ml^{-1}) carrier-free acid-free ^{32}P-orthophosphate (New England Nuclear) was added to the swirled sample. Aliquots (1-ml) were removed by an automatic pipette affixed with a sterile polyethylene top at timed intervals of 30, 100, 200, 300, 400, 500, and 1,000 sec after addition of the radiolabel. The 1-ml aliquots were filtered through presoaked Millipore HA filters at a pressure of 0.5 atm; then the filters were rinsed with three 10-ml rinses of distilled water. After the filters were air-dried, they were placed into Beckman Biogamma vials containing 2.5 ml Scintillene (Fisher Scientific) liquid scintillation cocktail and counted in a Beckman 6800 liquid scintillation spectrometer. The detected count rate was automatically corrected for quench, so differences in quenching from one sample to another were not a factor in the reported count rate (disintegrations per minute). Total radioactivity per millilitre was determined by placing 1.0 ml of the radioactive lake water into a 7-ml Beckman RediSolv MP water-accepting liquid scintillation cocktail.

18. The rate of uptake was determined from the regression $\ln (P_o/P_o - X)$ vs. time, where P_o was the total amount of ^{32}P added, X was the amount adhering to Millipore HA filters, and time was the time that the aliquot was filtered. Invariably, this yielded a line passing through the origin; the slope of this line, determined by linear regression, was the proportion k of available phosphate taken up per unit time (in units of hr^{-1}) (i.e., proportion of total phosphate remaining available transferred from solution to particles per hour).

19. The absolute rate of phosphate uptake was calculated as

$$v_{uptake} = k(SRP)$$

and was expressed as nmol ℓ^{-1} hr^{-1}.

Results

20. To meet the goals of this study, the P-dynamics of RBR were examined along a transect of the main stem during the time of maximum growth. The rate of uptake of phosphate and its rate of turnover were used as biochemical assays to determine whether phosphate availability was in growth-limiting supply. The contribution of DOP compounds in meeting the nutritional phosphate demand of the planktonic community was determined by comparing the rate of phosphate release from DOP with the rate of phosphate uptake by seston. Observations made at five stations along RBR were compared with the results of similar studies conducted at four stations along a main stem transect of CHL.

21. The overall results of this study indicate that P is a nutrient in growth-limiting supply and that DOP does not appear to satisfy a significant portion of the phosphate demand of plankton in RBR or CHL. DOP could serve as a significant P source if detrital decomposition in RBR should release substantial quantities of DOP as PME to the water, potentially affecting the communities in both RBR and CHL. Both RBR and CHL showed major differences between water sampled above and below the plunge point. Dynamics of P appeared to be similar in these two large reservoirs at this time.

Phosphate dynamics

22. Determining the rate at which dissolved phosphate moves to particles is fundamental to understanding the phosphorus metabolism of a lake. Phosphate is the only chemical form of P generally regarded to be directly assimilated by phytoplankton and bacterioplankton (Fogg 1974). Because of its unique metabolic role, determining the rate at which phosphate is taken up by plankton is a direct measure of the rate at which phosphorus becomes available to the planktonic community as a whole. The nutritional usefulness of other P-compounds depends on the rate at which phosphate is released from them. If a substance is a significant source of phosphate to the plankton, it is because the rate at which phosphate is released from that substance represents a substantial fraction of the rate at which phosphate is assimilated by the

plankton. Conversely, if the rate of release of phosphate from DOP is slow relative to the velocity of phosphate uptake, DOP would be judged an insignificant source of P to the planktonic community.

23. Besides its usefulness as a means for assessing the relative contribution of various compounds that release phosphate, phosphate dynamics are also useful in determining whether the plankton behave metabolically as if growing under P-limiting conditions. Lean et al. (1983) have shown that plankton under P-limitation assimilate phosphate rapidly with a turnover time of 20 min or less. Plankton under P-sufficient conditions slowly assimilate phosphate, with a turnover time of 1 hr or more. Although algal culture bioassays, such as those of Schelske (1984), often are used to determine whether natural populations are growth-limited by P availability, biochemical bioassays, such as phosphate dynamics, are preferable because they are quicker, cheaper, provide quantitative data regarding metabolism of the plankton, and are less susceptible to time-dependent artifacts (e.g., release of phosphate from complex P-compounds) (Cowen and Lee 1976).

Kinetics of phosphate uptake by seston

24. Uptake rates can be calculated if the kinetic dependence on phosphate can be determined and the phosphate concentrations present in the water are known. The kinetics of phosphate uptake were determined by adding tracer quantities of radiolabeled orthophosphate to surface water samples that were assumed to be in a dynamic equilibrium. Generally, in such radiotracer kinetic experiments, aliquots of the radiolabeled sample are filtered and the radioactivity of a portion of the filtrate is determined. Nalewajko and Lean (1980) have stated that this method suffers from the difficulty that some of the labeled material passing through the filter may be DOP released from particles. In such a case, the assumption that ^{32}P represents exclusively the available phosphate would lead to an underestimate of the proportional uptake constant. Also, the kinetics determined by such measurements would be confounded by the additional kinetics of DOP release. An alternate, somewhat more direct method follows the uptake of radiotracer into

seston by collecting each timed aliquot onto a separate filter, then washing and drying the filter and counting the radioactivity retained on the filter. A drawback of this method is that phosphate and other dissolved compounds may artifactually bind to the filter and thereby elevate the estimate of the proportional uptake rate constant. Furthermore, to interpret the results in a straightforward fashion, it must be shown that the total amount of tracer per aliquot is constant and that tracer is not being lost to compartments other than the suspended particles (e.g., to the walls of the flask). Transfer of radioactive phosphate (carrier-free $^{32}PO_4^{3-}$) into seston was followed by collecting particles from 1-ml aliquots on Millipore HA filters (0.45 μm average pore size) at timed intervals. Millipore HA filters retained more labeled material than Whatman GF/A (3.0 μm), Whatman GF/F (1.2 μm), and Nucleopore 0.6 μm. Also, Millipore GS (0.22 μm) and Gelman Metricel GA-8 (0.20 μm) and Nucleopore 0.2-μm filters retained no more label than did Millipore HA; in fact, Nucleopore 0.2-μm filters often retained somewhat less.

25. Addition of radiotracer to lake water prefiltered through Millipore HA filters showed that artifactual binding of nonparticulate radioisotope was virtually eliminated by presoaking the filters in 0.5 mol KH_2PO_4. Only 90 ±15 SD dpm were retained on presoaked filters through which was passed 2.7×10^4 dpm in 1.0-ml prefiltered lake water. The total amount of radiotracer detected remained unchanged over the duration of the test, indicating that none was lost to the sides of the flask, altering its availability to suspended particles. For these reasons, the radioactive material retained on the filters is considered an accurate indication of ^{32}P-phosphate that had moved from solution to suspended particles.

26. Some investigators have shown that laboratory and field populations can take up phosphate at a rate having a simple first-order (linear) dependence on phosphate concentration (Rhee 1972, 1973; Perry 1976). Others have shown in laboratory studies that uptake can be complex, apparently proceeding by several pathways (Azad and Borchardt 1970, Chisholm and Stross 1976) that may result in "nonlinear" kinetics.

Some uptake may be enzyme-mediated and may require metabolic energy, exhibiting Michaelis-Menten kinetics; some uptake may occur through facilitated diffusion or passive sorption to surfaces and may exhibit strict first-order kinetics. Lean (1976) and Twinch and Breen (1984) showed that natural populations may exhibit "biphasic" kinetics, and that the kinetics of uptake may vary seasonally and with nutrient status of the water.

27. If uptake exhibits a first-order dependence on phosphate remaining available in solution, the uptake velocity (v) would be

$$v = \frac{dx}{dt} = k(P) \qquad (1)$$

where x is the amount taken up by particles at time t, k is the proportion of P taken up per unit time, and P is that portion of the initial amount of dissolved phosphate (P_o) not yet taken up by particles:

$$P = P_o - x$$

Equation 1 becomes:

$$\frac{dx}{dt} = k(P_o - x)$$

Its integrated form is:

$$\ln \frac{P_o}{P_o - x} = kt \qquad (2)$$

Plotting values of $\ln(P_o/P_o - x)$ against t gives a straight line passing through the origin. The slope of this line, k, is the proportional rate constant; its inverse is the "turnover time" T, the

interval required to remove all of the material from solution, if it moved constantly at the initial rate.

28. The overall first-order dependence on available phosphate is apparent from the straight line passing through the origin (Figure II-1). This initial slope was determined by linear regression of those values observed in the first several hundred seconds of the test. The regression coefficient was consistently high; r^2 was generally greater than 0.99. Replicate determinations yielded lines with similar initial slopes generally within a standard deviation of 3 percent of the mean slope. This initial slope, the overall rate constant k, is the proportion of available phosphate removed per unit time by all first-order processes. Usually the uptake showed an abrupt change after several hundred seconds, yielding a second line with a lower slope, indicating that uptake was not accomplished by a single first-order process but rather by two or more processes, each of which appeared to be first-order dependent on phosphate. Using this overall proportional uptake constant, the velocity of transfer of phosphate from solution to particles was calculated by Equation 1, estimating the phosphate concentration by the molybdenum-blue colorimetric method of Murphy and Riley (1962).

29. Observed turnover times indicated that, in both reservoirs, the plunge point discontinuously delimited the surface waters into two classes of seston. Figure II-1 and Table II-1 show that, in RBR, phosphate uptake was slow and the turnover time was very long above the plunge point (between Stations 190 and 180 on 19 July and Stations 180 and 160 on 8 August). Phosphate uptake was rapid and the turnover time was very short below the plunge point with no apparent gradient. Table II-2 shows that a similar pattern existed in CHL but to a less marked degree. The turnover time expressed the steady-state time necessary to move all available phosphate from solution to seston, and so represented the relative rate at which phosphate moved. A long turnover time implies a slow rate of phosphate processing; a short turnover time indicates rapid processing of available phosphate and is a biochemical indicator of P-limitation. Above the plunge point, the relative rate of

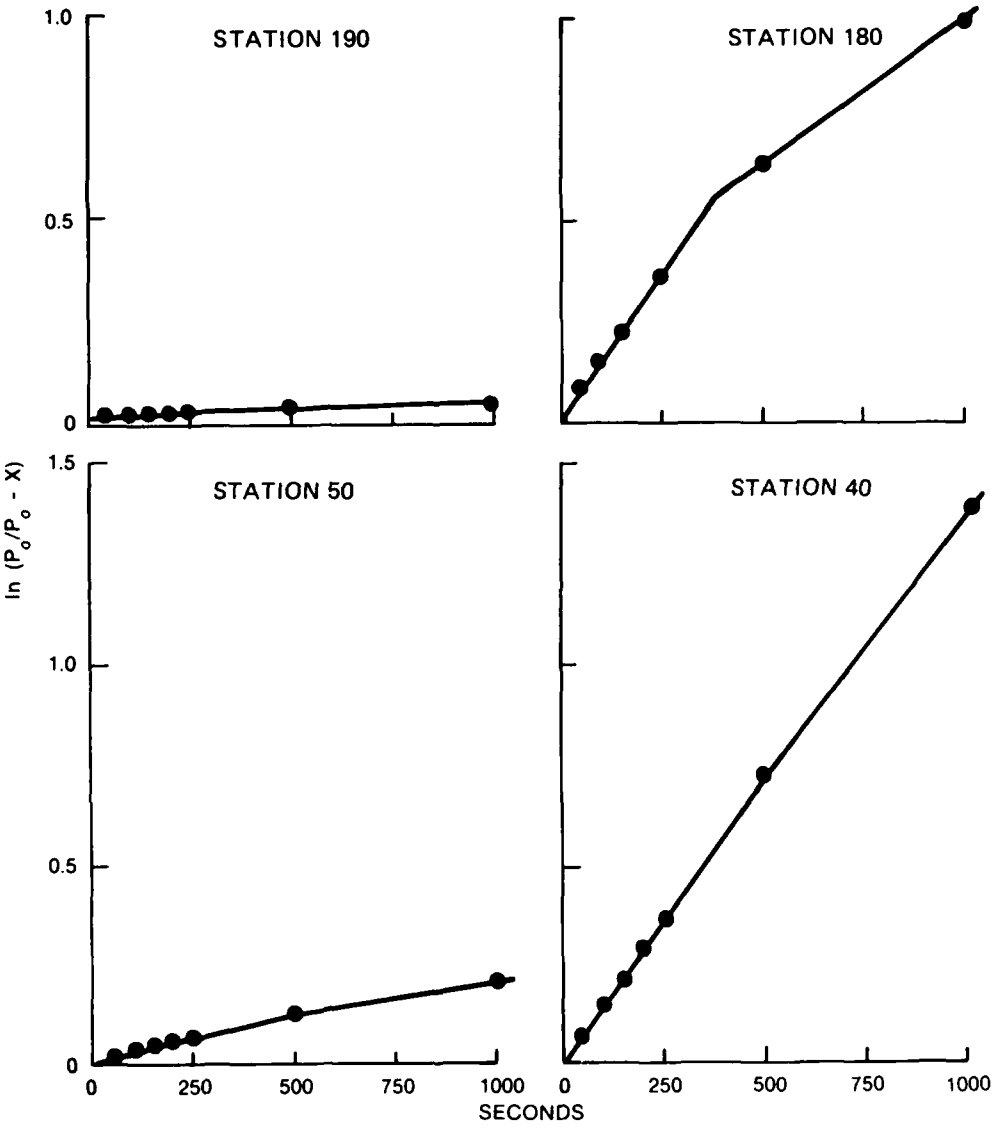

Figure II-1. Phosphorus uptake by seston as a function of time for water samples from selected stations. Samples were collected on 19 July 1984

Table II-1

Phosphate Dynamics - Richard B. Russell Lake

Date	Site	T min	k Total hr^{-1}	Correlation Coefficient r^2	SRP-P $\mu g\ \ell^{-1}$	SRP-P $nmol\ \ell^{-1}$	Velocity of PO_4 Uptake $nmol\ \ell^{-1} hr^{-1}$
19 Jul 84	190	360.8	0.166	0.998	24.18	780.0	129.5
	180	11.8	6.076	0.999	2.39	77.1	391.4
	160	8.6	6.997	0.999	1.27	41.0	286.0
	120	10.3	5.810	0.999	3.14	101.3	588.6
	60	10.7	5.591	0.996	3.69	119.0	665.3
8 Aug 84	190	372.0	0.161	0.997	1.32	42.6	6.9
	180	185.8	0.323	0.995	1.88	60.6	19.6
	160	13.8	4.347	0.999	1.32	42.6	185.2
	120	12.5	4.801	0.999	0.56	18.1	86.9
	60	12.0	5.007	0.999	0.10	3.2	16.0

Table II-2

Phosphate Dynamics - Clarks Hill Lake

Date	Site	T min	k Total hr^{-1}	Correlation Coefficient r^2	SRP-P $\mu g\ \ell^{-1}$	SRP-P nmol ℓ^{-1}	Velocity of PO_4 Uptake nmol $\ell^{-1} hr^{-1}$
19 Jul 84	50	71.0	0.846	0.996	3.14	101.3	85.7
	40	11.8	5.065	0.999	3.14	101.3	513.1
8 Aug 84	50	22.5	2.665	0.995	0.10	3.2	8.5
	40	12.9	4.639	0.997	1.88	60.6	281.1
	30	9.4	6.351	0.999	0.10	3.2	20.3
	20	11.7	5.126	0.999	1.88	60.6	310.6

phosphate movement is slow--indicative of P-sufficient conditions. Below the plunge point the turnover times were consistently on the order of 10 min, indicative of P-limitation.

30. The turnover time is only a relative indicator and so may provide an incorrect interpretation if used as an indicator of actual uptake rates. The actual uptake rate was calculated by multiplying the proportional uptake constant by the estimate of phosphate available in solution (SRP). The proportional uptake rate (k_{total}) was determined as the slope of the initial points forming a straight line passing through the origin; the slope of this line was estimated by linear regression. The correlation coefficient (r^2) was consistently greater than 0.99, indicating consistent reliability of the data and their interpretation as indicative of first-order dependence on phosphate concentrations. The SRP concentration, expressed in nmol ℓ^{-1}, was used to give an overall uptake velocity in nmol ℓ^{-1} hr^{-1}.

31. In both reservoirs the actual velocity of phosphate uptake by seston (both living and nonliving particles) increased greatly at the plunge point, increasing from 4 to 35 times. Although this qualitatively is a similar result to that observed for turnover time, Figure II-2 shows that proportional uptake rate (k) and actual velocity (v) did not necessarily coincide. The proportional uptake rates at Stations 160 and 60 on RBR on 8 August were similar, but the actual rate of transfer of phosphate to particles was 10-fold greater at Station 160. Conversely, the actual rate of transfer at Stations 180 and 60 were similar, but the turnover time was 15 times longer at Station 180 because there was considerably more phosphate available above the plunge point at Station 180 than in the nutrient-poor lower basin.

32. Taken together, these results indicate that, above the plunge points, both the relative and actual velocity of phosphate uptake are slow, indicative of P-sufficient conditions. At the plunge point, available phosphate is supplied and rapidly taken up by P-limited seston. Further down the reservoir the actual rate of phosphate uptake may slow, not because of a change in sestonic demand (i.e., demand

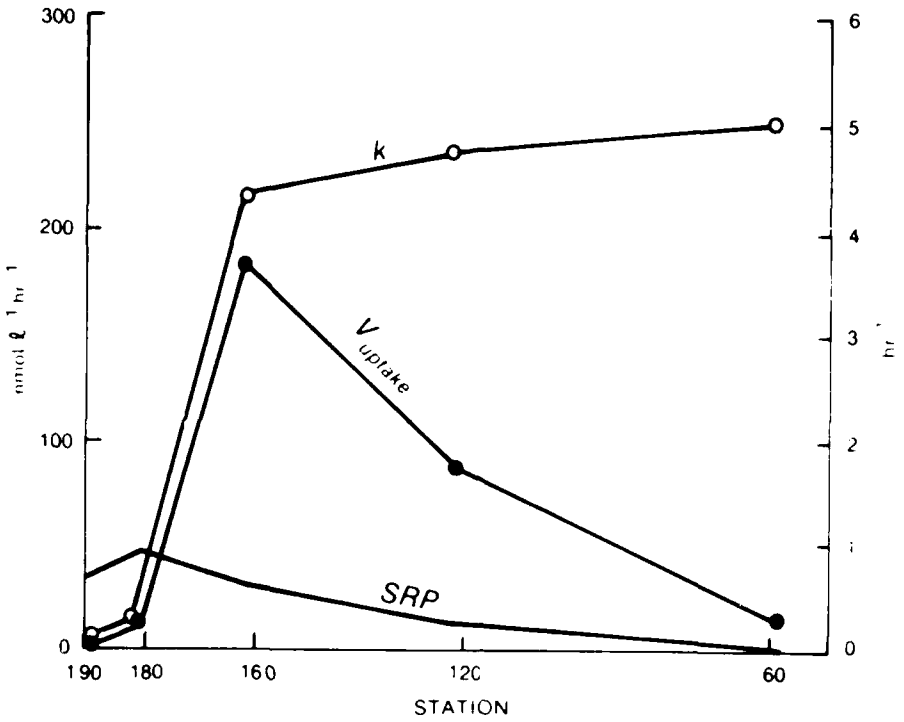

Figure II-2. Between-station changes in proportional (k) and actual (V_{uptake}) phosphorate uptake rate for seston, and soluble reactive phosphorus (SRP) concentration in RBR, 8 August 1984

remained high as shown by a high proportional uptake rate) but because of decreased amounts of available phosphate (Figure II-2).

Active and inactive phosphate uptake

33. Removal of phosphate from solution by particles can occur by active or passive processes, including sorption to surfaces by anion exchange, passive diffusion, facilitated diffusion into cells, and ATP-dependent "active" uptake into living cells. Understanding the processes involved in the turnover of phosphate depends on discriminating between these various uptake mechanisms. To discriminate between "active" and passive processes, 1×10^{-4} mol potassium cyanide (CN) was added to whole lake water samples 30 min before addition of ^{32}P-phosphate. Elsewhere (Heath 1986) we have shown that this interval of pretreatment is sufficient to diminish the rate of uptake to a

cyanide-resistant rate of phosphate uptake; longer preexposure to CN did not alter this rate of uptake.

34. Phosphate uptake in each of the samples was studied in the absence and in the presence of CN; the data express proportional uptake rates under each condition. Phosphate uptake in the unpoisoned water was expressed as the "total uptake rate," k_{TOT}. Uptake in the CN-poisoned sample was called "cyanide-insensitive uptake," k_{CN}. The "active uptake," k_{ACT}, was calculated as the difference between the total uptake rate and the cyanide-insensitive uptake. It would be more correct to term this as the cyanide-sensitive uptake rate, avoiding the implication that "active" implies uptake by cells and "cyanide-insensitive" uptake implies sorption to nonliving surfaces.

35. The term active is used here with the caveat that numerous investigators have shown that phosphate uptake by different species of algae in axenic cultures differ greatly in their sensitivity to cyanide. Gest and Kamen (1948) showed that phosphate uptake by *Chlorella pyrenoidosa* and *Scenedesmus* sp. was completely insensitive to CN; Rivkin and Swift (1982) showed that phosphate uptake by *Pyrocystis noctiluca* was greatly diminished by 1×10^{-4} mol CN; Chisholm and Stross (1976) showed that CN completely inhibited phosphate uptake by *Euglena gracilis*.

36. The pattern of cyanide-insensitive uptake differed from that of the active uptake in each reservoir. Typically, both active and CN-insensitive inactive proportional uptake increased abruptly at the plunge point, but the active proportional uptake remained relatively constant while the CN-sensitive inactive proportional uptake reached a maximum below the plunge point and then declined toward the dam station (Figure II-3). Table II-3 presents the actual rate of uptake due to CN-insensitive and active uptake in RBR; Table II-4 shows the same for CHL. These calculations of the actual phosphate uptake rates by active and inactive processes again showed no parallel with the rate of proportional uptake, because of their dependence on the spatially varying phosphate concentrations in situ. The only pattern seen in both reservoirs was an abrupt change in the actual rate of transfer to particles by both active and inactive processes.

Table II-3

Cyanide-Sensitive (Active) and Cyanide-Insensitive (Inactive)

Phosphate Uptake - Richard B. Russell Lake

Date	Station	k total hr^{-1}	k CN-Insensitive hr^{-1}	k CN-Sensitive hr^{-1}	Velocity Inactive PO_4 Sorption nmol $\ell^{-1} hr^{-1}$	Velocity Active PO_4 Uptake nmol ℓ^{-1} hr^{-1}	Active Total
19 Jul 84	190	0.166	0.050	0.116	39.0	90.5	0.70
	180	5.076	0.439	4.637	33.8	357.5	0.91
	160	6.977	1.285	5.692	52.7	233.4	0.82
	120	5.810	1.123	4.687	113.8	474.8	0.81
	60	5.591	0.506	5.084	60.2	605.0	0.91
8 Aug 84	190	0.161	*0.100*	0.061	4.3	2.6	0.38
	180	0.323	0.095	0.228	5.7	13.8	0.71
	160	4.347	2.100	2.247	89.5	95.7	0.51
	120	4.801	1.673	3.128	30.3	56.6	0.65
	60	5.007	1.653	3.354	5.3	10.7	0.67

Table II-4

Cyanide-Sensitive (Active) and Cyanide-Insensitive (Inactive)

Phosphate Uptake - Clarks Hill Lake

Date	Station	k total hr^{-1}	k CN-Insensitive hr^{-1}	k CN-Sensitive hr^{-1}	Velocity Inactive PO_4 Sorption nmol $\ell^{-1} hr^{-1}$	Velocity Active PO_4 Uptake nmol ℓ^{-1} hr^{-1}	Active Total
19 Jul 84	50	0.846	0.088	0.758	8.9	76.8	0.90
	40	5.065	1.433	3.632	145.2	367.9	0.72
8 Aug 84	50	2.665	0.346	2.319	1.1	0.4	0.87
	40	4.639	0.300	4.339	18.2	262.9	0.93
	30	6.351	1.777	4.574	5.0	14.6	0.72
	20	5.126	1.116	4.010	67.6	243.0	0.78

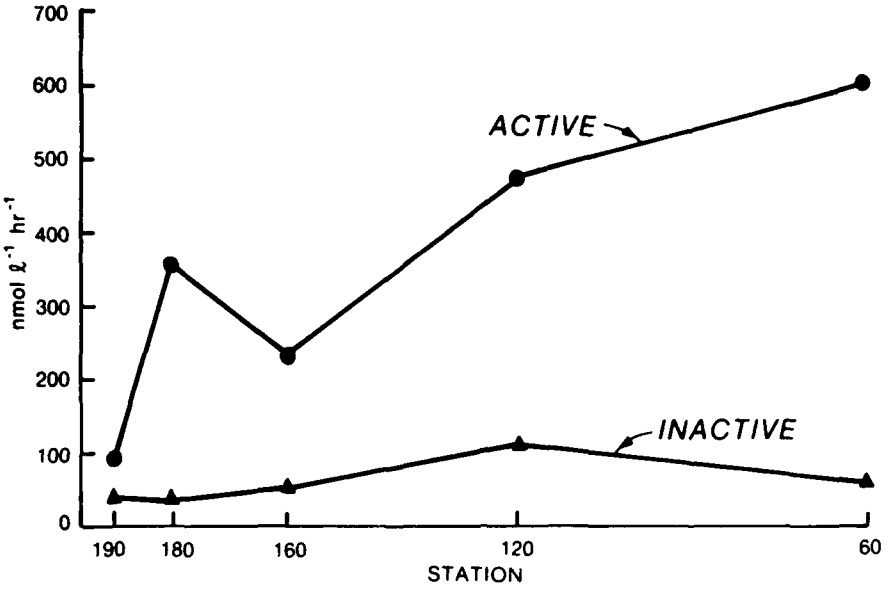

Figure II-3. Between-station differences in active and inactive phosphate uptake by seston in Richard B. Russell Lake on 19 July 1984

37. Contrary to expectations, the portion of uptake due to active CN-sensitive processes was generally high and did not change in any consistent pattern along the length of the reservoir. It was reasonable to expect that nutrient-rich input waters would have had relatively little plankton and so the uptake above the plunge point would be dominated by inactive, CN-insensitive processes, while below the plunge point the uptake would be dominated by active uptake. The closest our observations came to meeting this expectation was on 8 August in RBR when uptake was dominated by inactive processes at Station 190, but was dominated by active processes at all lower sites. In RBR on 19 July, active uptake predominated at all sites, but it was lowest above the plunge point. In contrast to the observations for RBR, in CHL, active uptake consistently represented between 70 and 90 percent of the uptake of phosphate at all stations.

38. As a means of investigating the possibility of progressive alterations in the pattern of P-availability along the length of the reservoir, the velocity of uptake was scaled to the particulate-P (total P minus total soluble P) and also was scaled to active chlorophyll *a*

(total chlorophyll minus phaeopigments). The amount of particulate-P per the amount of active chlorophyll should be high when many inorganic particles are present and when living algal cells have large amounts of phosphorus storage compounds (i.e., when grown under P-sufficient conditions). Olsen, Knutsen, and Lien (1983) have shown that when algal cells were cultured axenically under P-limitation, the P-content declined and approached levels as low as 12 nmol P per µg chlorophyll a. In RBR the amount of particulate-P per unit amount of chlorophyll decreased progressively down the reservoir (Figure II-4, Table II-5), indicating a progressively P-deficient phytoplankton community. In contrast with this finding, CHL showed the least amount of particulate-P per unit chlorophyll at the uppermost station (Station 50), with the value of the quantity nmol P per µg chl a tending to increase down the length of the reservoir (Figure II-5, Table II-6).

39. Profiles of velocity of active phosphate uptake per unit amount of particulate-P and the velocity of phosphate uptake per unit amount of active chlorophyll a showed similar patterns in both reservoirs (Figures II-4 and II-5, Tables II-5 and II-6). The uptake rate scaled for the amount of particulate-P estimated the turnover of phosphorus through particles (i.e., the proportion of particulate-P actively taken up per hour), a measure of intrinsic assimilation. In each reservoir there was an abrupt increase in the active turnover of particulate-P at the plunge point, with a pronounced tendency for increases toward the dam station. Especially on RBR on 19 July, the active turnover of particulate-P at Stations 120 and 60 (5.652 and 6.237 hr, respectively) indicated that particulate-P was turned over (completely replaced) several times per hour. Such high assimilation rates imply extreme P-limitation (Lean et al. 1983). Although a similar pattern of increased turnovers of particulate-P down the length of CHL was observed, it was not as pronounced. Also, on 19 July, the greatest intrinsic turnover was observed above rather that below the plunge point. On 8 August, the fastest (i.e., shortest) turnover times of particulate-P were about 75 min on each reservoir. Also indicative of a progressively P-limited phytoplankton was the observation of continual

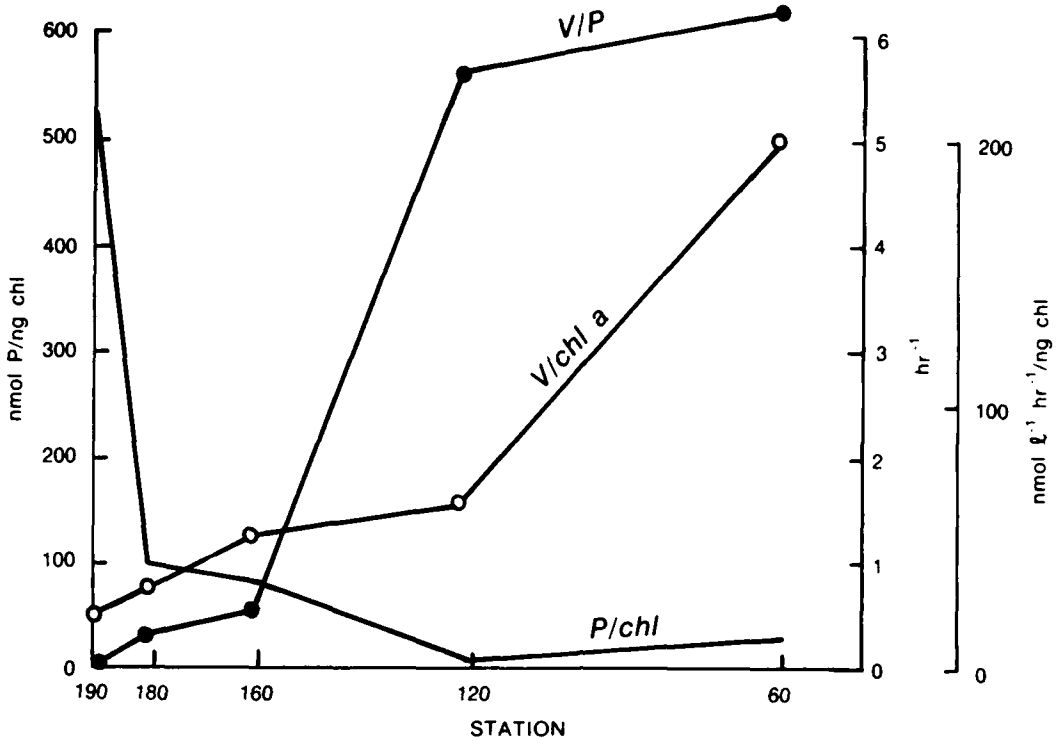

Figure II-4. Between-station differences in the rate of phosphate uptake per unit of particulate phosphorus (V/P; hr^{-1}), the rate of phosphate uptake per unit of chlorophyll (V/chl a; nmol ℓ^{-1}/ng chl), and the amount of phosphate per unit of chlorophyll a (P/chl; nmol P/ng chl) in Richard B. Russell Lake on 19 July 1984

increase in active phosphorus uptake per unit amount of chlorophyll at various stations down the length of each reservoir.

40. These findings taken together indicate that P was an important limiting resource in RBR and CHL. In each, phosphate availability appeared to limit planktonic community activities. Using phosphate uptake rate as a bioassay indicator of the phosphorus nutritional status of the plankton, P-limitation appeared to increase progressively down the length of each reservoir. In RBR these tendencies were more pronounced than in CHL; only in RBR was extreme biochemical P-limitation observed. Each reservoir showed marked differences above and below the plunge point, but these differences were much more pronounced on RBR.

Table II-5

Active Uptake Versus Particulate P and Chlorophyll, Richard B. Russell Lake

Date	Station	Total P $\mu g\ \ell^{-1}$	Total Sol.* P $\mu g\ \ell^{-1}$	Total Part. P** nmol ℓ^{-1}	Chl a $\mu g\ \ell^{-1}$	nmol Part. P/ μg Chl a	nmol ℓ^{-1} hr^{-1}	V_{Act}† Part. P hr^{-1}	V_{Act} Chl a/ nmol P $\mu g\ Chl^{-1}\ hr^{-1}$
19 Jul 84	190	100.3	23.6	2,474	4.149	551.0	90.5	0.036	20.16
	180	49.9	13.2	1,184	11.975	98.9	357.5	0.302	29.85
	160	26.1	12.6	435	5.239	83.0	233.4	0.537	44.55
	120	19.6	17.0	84	7.484	11.2	474.8	5.652	63.44
	60	17.7	14.7	97	2.994	32.4	605.0	6.237	202.07
8 Aug 84	190	14.7	15.4	474††	0.924‡	513.0	2.6.	0.005	2.81
	180	17.3	11.1	200	1.497	133.6	13.8	0.069	9.22
	160	28.3	12.9	497	5.239	94.9	95.7	0.193	18.27
	120	15.7	13.5	71	2.245	31.6	56.6	0.797	25.21
	60	17.3	15.5	58	5.239	11.1	10.7	0.184	2.04

* Soluble phosphorus.
** Particulate phosphorus.
† Velocity of active substance.
†† Total P.
‡ Total chl a.

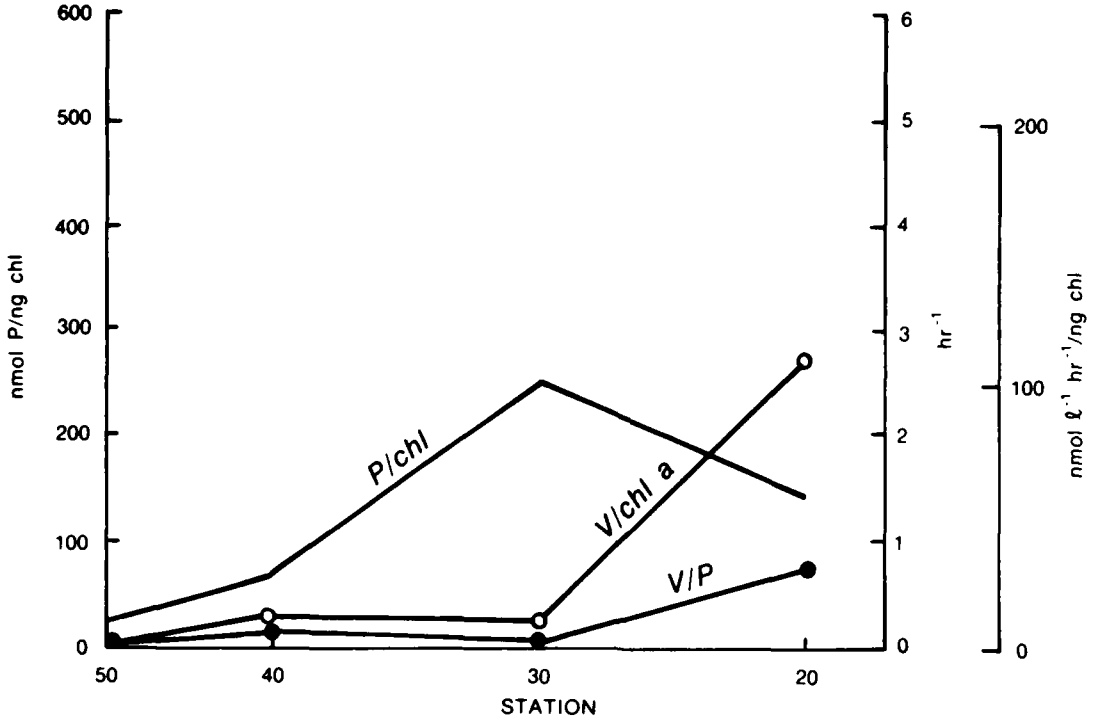

Figure II-5. Between-station differences in the rate of phosphate uptake per unit of particulate phosphorus (V/P; hr^{-1}), the rate of phosphate uptake per unit of chlorophyll (V/chl a; nmol ℓ^{-1}/ng chl), and the amount of phosphate per unit of chlorophyll a (P/chl; nmol P/ng chl) in Clarks Hill Lake on 8 August 1984

Dissolved organic
phosphorus compounds

41. Whether DOP compounds are significant sources of phosphate for freshwater plankton is unknown. Hutchinson (1957) noted that DOP compounds often occurred in abundance relative to barely detectable quantities of phosphate at times of maximum phytoplankton productivity. From this he inferred that DOP may be an important autochthonous source of phosphate. Conversely it can be reasoned that such compounds are abundant because they are relatively refractory and are not significantly utilized as a phosphate source. Rigler (1968) and Berman (1970) observed substantial portions of the "soluble unreactive phosphorus" (SUP) fraction of compounds that appeared to be refractory to phosphate release. Francko and Heath (1979) demonstrated that phosphate release

Table II-6

Active Uptake Versus Particulate P and Chlorophyll, Clarks Hill Lake

Date	Station	Total P $\mu g\ \ell^{-1}$	Total Sol. P $\mu g\ \ell^{-1}$	Part. P $nmol\ \ell^{-1}$	Chl a $\mu g\ \ell^{-1}$	nmol Part. P/ μg Chl a	$nmol\ \ell^{-1}\ hr^{-1}$	V_{Act} Part. P hr^{-1}	V_{Act} Chl a/ nmol P $\mu g\ Chl^{-1}\ hr^{-1}$
19 Jul 84	50	17.4	15.6	58	5.239	11.1	76.8	1.324	14.66
	40	30.6	19.2	368	7.484	49.2	367.9	1.000	49.16
8 Aug 84	50	21.4	16.5	158	5.987	26.4	7.4	0.047	1.24
	40	65.9	71.3	1,568	23.201	67.6	262.9	0.168	11.33
	30	21.9	10.3	374	1.497	249.8	14.6	0.039	9.75
	20	14.7	4.5	329	2.245	146.5	243.0	0.739	108.24

from DOP is dependent on at least two distinct processes, and that the conditions essential for release of phosphate from DOP may not always be present in freshwater systems. DOP compounds can be significant sources of P only if phosphate is released from them at a rate comparable to the planktonic assimilation rate of available phosphate.

42. Nutritionally significant DOP compounds are those that release phosphate under natural conditions. Determining their significance depends on identifying the manner by which phosphate is released. Rather than classifying DOP compounds structurally, we have classified them functionally according to the mechanisms involved in phosphate release. Francko and Heath (1979) reported the release of phosphate from two different classes of dissolved organic compounds. One of these classes had a high molecular weight and released phosphate following irradiation with sunlight in an event coincident with the photoreduction of ferric ions associated with these compounds (Francko and Heath 1982, 1983). These compounds co-chromatographed with humic acids both in Sephadex and in anion-exchange chromatography. Also, they are more frequently seen in water from acid humic bog lakes (Cotner 1984). For these reasons, we believe that they are iron-containing humic acids to which phosphate is bound. These compounds are detected by increased SRP during irradiation with a low-intensity ultraviolet (UV) lamp (Francko and Heath 1979, Cotner 1984). The rate of phosphate release from these humic compounds appeared to be limited by a process involving bound-ferric ion availability to an electron donor within the compound. Rate did not appear to be dependent on the intensity of the UV light supplied (Francko and Heath 1982).

43. No photosensitive release of phosphate was detected in either reservoir at any station. Many aromatic compounds can be detected by a characteristic absorbance of light having a wavelength of 250 nm (A_{250}); also, many compounds fluoresce. Characteristically, humic compounds both absorb UV light and fluoresce (400 nm excitation; 530 nm emission). We observed a twofold increase in A_{250} of filtered water at the plunge point in RBR, with gradual decline toward the dam station (Figure II-6). No significant amount of fluorescence was noted in RBR; also, no trends

in change of fluorescence were observed down the length of RBR. The only significant absorbance or fluorescence noted in either reservoir was seen at Station 40 in CHL (Figure II-6). This corresponded with a major phytoplankton bloom, and so probably represented phaeopigment released into the water by dead algal cells. Aside from this station, the absorbance and fluorescence of filtered water in CHL were similar to RBR, showing no significant trends along the length of the reservoir.

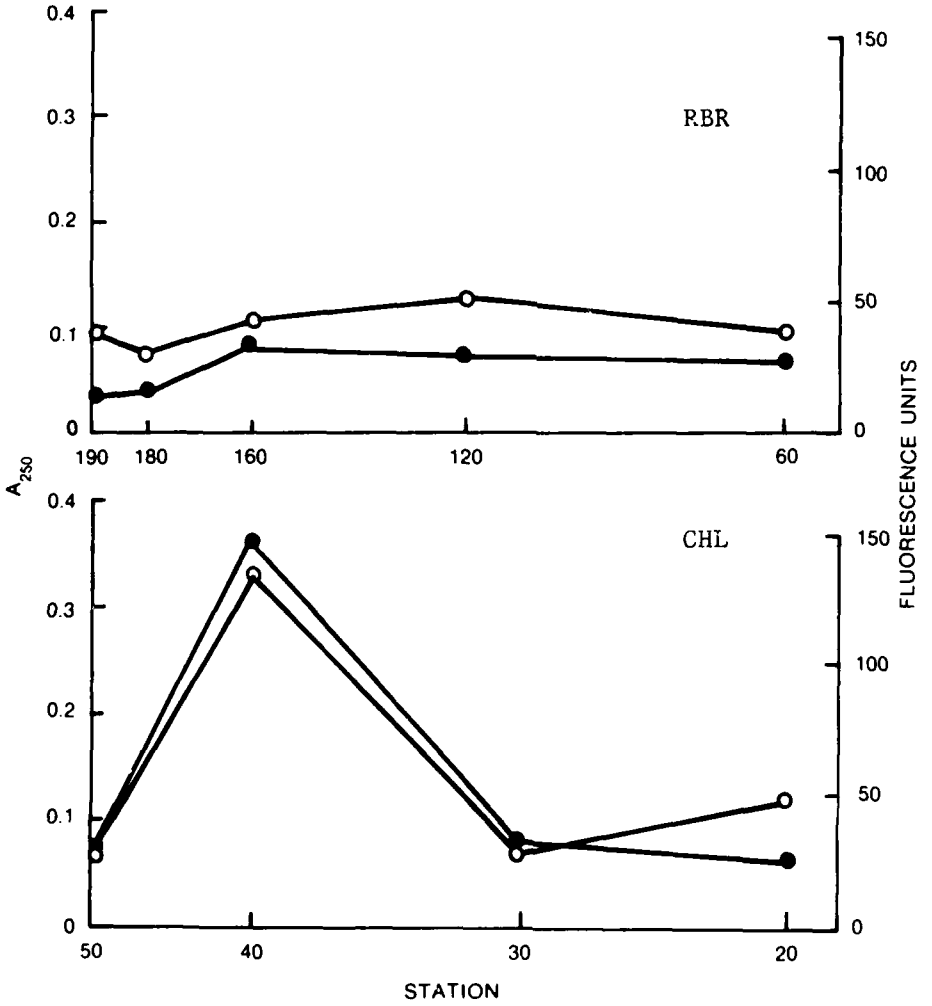

Figure II-6. Between-station differences in absorbance of light at a wavelength of 250 nm (open circles) and fluorescence (closed circles) for water samples collected on 8 August 1984 in Richard B. Russell Lake (upper) and Clarks Hill Lake (lower)

44. A second class of complex phosphorus compounds observed by Francko and Heath (1979) are phosphomonoesters (PME) that release phosphate, through the hydrolytic action of phosphatases. These phosphatases are present both as particulate exoenzymes attached to algal and bacterial cells (Kuenzler 1970) and as dissolved enzymes that are apparently released by zooplankton (Jansson 1976, Boavida and Heath 1984). Phosphatases in general have a broad substrate specificity (Fernley 1971) and conceivably could release phosphate from a wide variety of compounds. Some phosphatases can release phosphate only from PME, while other phosphatases are capable of hydrolyzing terminal phosphoryl groups from both PME and polyphosphates.

45. Little is known about the structure, origin, and fate of naturally occurring phosphatase-sensitive compounds. We have shown that these compounds have a low molecular weight of generally less than 1,000 daltons (Francko and Heath 1979) and vary seasonally (Heath and Cooke 1975, Boavida 1984, Cotner 1984). Kuenzler (1970) observed that PME released by algal cells are not necessarily substrates for the photophatases produced by those cells. It is generally recognized that PME originate from detrital decomposition; whether they are released by "healthy" cells is a subject of controversy (Fogg 1977; Sharp 1977, 1978; Aaronson 1978).

46. The concentrations of naturally occurring PME substrates were determined as the increase in detectable SRP in filtered lake water after incubation with calf intestinal mucosa alkaline phosphatase. This enzyme hydrolyzes a broad variety of PME as well as certain polyphosphates (Fernley and Walker 1967). Because not all phosphatases accept as broad a range of substrates as this enzyme, this estimate of the concentration of phosphatase-sensitive compounds is an upper limit of the substrate concentration available to naturally occurring enzymes. Phosphomonoesters often comprised a major portion of the dissolved organic P, shown in Tables II-7 and II-8 as SUP (the difference between total soluble P and SRP). Especially on 8 August 1984, PME were detected at most stations sampled on each reservoir. PME concentrations tended to be low above the plunge point and to increase down the reservoir. On

Table II-7

Phosphomonoesters and Alkaline Phosphatase Activity

Richard B. Russell Lake

Date	Station	SUP	PME		V_{max}	K_m	$v_{release}$
		$\mu g\ \ell^{-1}$	$\mu g\ \ell^{-1}$	$nmol\ \ell^{-1}$	$nmol\ \ell^{-1}\ hr^{-1}$	$\mu mol\ \ell^{-1}$	$nmol\ \ell^{-1}\ hr^{-1}$
19 Jul 84	190	0.6	3.0	96.8	0	150	0.0
	180	10.8	0.0	0.0	1,646	150	0.0
	160	11.3	0.0	0.0	1,534	150	0.0
	120	13.8	0.0	0.0	1,937	150	0.0
	60	11.1	3.0	96.8	2,228	150	1.437
8 Aug 84	190	14.1	6.0	193.5	392	150	0.505
	180	9.2	1.1	35.5	403	150	0.095
	160	11.6	7.5	241.9	1,220	150	1.964
	120	14.9	15.4	496.8	1,970	150	6.503
	60	15.5	0.0	0.0	2,004	150	0.0

Table II-8

Phosphomonoesters and Alkaline Phosphatase Activity

Clarks Hill Lake

Date	Station	SUP	PME		V_{max}	K_m	$v_{release}$
		$\mu g\ \ell^{-1}$	$\mu g\ \ell^{-1}$	$nmol\ \ell^{-1}$	$nmol\ \ell^{-1}\ hr^{-1}$	$\mu mol\ \ell^{-1}$	$nmol\ \ell^{-1}\ hr^{-1}$
Jul 84	50	14.5	0.0	0.0	694	250	0.0
	40	16.0	0.0	0.0	2,586	250	0.0
8 Aug 84	50	16.5	0.3	9.7	940	250	0.036
	40	15.4	10.3	332.3	1,097	250	1.456
	30	10.3	4.5	145.2	2,071	250	1.202
	20	2.6	0.0	0.0	2,306	250	0.0

both RBR and CHL, the PME reached a maximum in the midpoint of the reservoir, declining toward the dam station.

47. The rate of release of phosphate from these phosphatase-sensitive compounds depends on the enzyme activity present in the lake water and the parameters controlling the kinetics of those enzymes. Phosphatases generally are not homotropically modulated by their substrates and conform to Michaelis-Menten kinetics (Fernley 1971). Although they often are product-inhibited by orthophosphate, the phosphate concentrations that appreciably affect enzyme activity (i.e., greater than 1×10^{-5}) (Gottesman, Simpson, and Vallee 1969) are at least two orders of magnitude greater than phosphate concentrations found in the lakes studied. The velocity of release of phosphate from PME can be estimated by calculation from the Michaelis-Menten equation following determination of the substrate concentrations naturally occurring in the lake water and determination of the collective maximal velocity (V_{max}) and the collective Michaelis constant (K_m) of the enzymes present in the water. The enzyme variables V_{max} and K_m of the collective activity in the lake were determined using p-nitrophenyl phosphate (pNPP) as the substrate over concentrations ranging from 2.5×10^{-5} mol to 2×10^{-3} mol pNPP. Linear regression of the reciprocal of initial velocity of hydrolysis versus the reciprocal of pNPP concentration yielded estimates of K_m and V_{max} (Lineweaver and Burk 1934). The regression coefficient was always high, generally $r^2 > 0.98$, providing a reliable estimate of these variables.

48. The use of p-nitrophenyl phosphate (pNPP) as a model substrate provided a reasonable estimate of phosphatase activity. Bacterial alkaline phosphatase (Reid and Wilson 1971) and mammalian alkaline phosphatases (Fernley 1971) appear to accept pNPP and natural substrates equally well (i.e., the K_m values for pNPP and natural substrates are similar). Acid phosphatases often hydrolyze pNPP somewhat faster than natural substrates (Hollander 1971). This artificial substrate has been shown in some cases to be a reliable model substrate for the investigation of algal phosphatases (Flint and Hopton 1977), although this has

not yet been demonstrated in a wide variety of eukaryotic algae (Cemella, Antia, and Harrison 1983).

49. The progress of this reaction is followed by detecting p-nitrophenol released on hydrolysis, so natural substrates act as competitive inhibitors of this model reaction, decreasing the rate of hydrolysis of pNPP by competing with it for the enzyme. Competitive inhibitors increase the apparent K_m for pNPP by a factor of $(1 + I/K_I)$, where I is the concentration of natural substrates and K_I is the Michaelis constant of the enzyme for these natural substrates (Cornish-Bowden 1979). The presence of natural substrates caused only a slight error (less than 10 percent) in the estimate of the K_m for pNPP, because I was usually less than 1×10^{-6} mol and K_I appeared to be greater than 1×10^{-5} mol.

50. Alkaline phosphatase activity in RBR and CHL (reported as V_{max}) consistently increased abruptly at the plunge point and continued to increase gradually along the length of the reservoir. The velocity of release of phosphate from PME was a function of both enzyme activity (V_{max}), enzyme affinity for the PME substrate (K_m), and the substrate concentration (PME) as expressed by the Michaelis-Menten equation:

$$v_{rel} = \frac{V_{max}\,(PME)}{K_m + (PME)} \qquad (3)$$

The rate of release of phosphate from PME was relatively slow at all stations observed. It tended to reach a maximum toward the midregions of each reservoir. The portion of PME turned over per unit time (i.e., v_{rel} per PME) was always very slow.

51. The PME did not appear to serve as nutritionally significant sources of phosphorus to planktonic organisms. The turnover rate (the proportion of PME turned over per unit time) was always very slow. The greatest turnover rate observed was 0.015 hr^{-1}, implying that the shortest turnover time of the PME pool was about 67 hr long. More directly, the fraction of total sestonic phosphate uptake satisfied by release of

phosphate from PME was calculated as v_{rel} per v_{uptake}. These studies indicated that phosphate released from PME never satisfied more than 7.4 percent of the sestonic phosphate demand, and generally satisfied less than 1 percent of the phosphate demand.

52. Despite these findings that PME were not significant sources of P, the phosphatase activities observed are useful indicators of the nutritional status of the planktonic community. The enzyme specific activity was determined by dividing maximum possible velocity (V_{max}) by particulate-P and dividing V_{max} by chlorophyll a (Tables II-9 and II-10). Fitzgerald and Nelson (1966), Jones (1972), and Heath and Cooke (1975) have noted that as algae and/or bacteria become more P-limited, the specific activity of alkaline phosphatase associated with these cells increases. The phosphatase specific activity increased from barely detectable levels above the plunge point to very high specific activities toward the dam station in RBR. The specific activities in CHL were sufficiently high to indicate P-limited plankton throughout the reservoir. In RBR, only below the plunge point did plankton appear to be P-limited as determined by this biochemical bioassay.

Discussion

53. The results of this study lead to several conclusions that imply topics for future investigation. These conclusions are also useful in developing a watershed management plan.

54. Planktonic communities in RBR and CHL were P-limited. Considerable chemical and biochemical evidence supports the view that the plankton in each of these reservoirs were growth-limited by P-availability during July and August at the sampled stations. The quantity of phosphate, detected as SRP, reached very low concentrations (i.e., less than 1×10^{-7} mol) late in the summer. This low phosphate availability influenced the nutritional status of the plankton, reflected in their biochemical activities. Phosphate uptake increased to very high rates and the phosphate turnover time decreased to low values that are characteristic of P-limited communities (Lean et al.

Table II-9

Significance of Phosphatase Release of Phosphate from PME

Richard B. Russell Lake

Date	Station	$\frac{v_{rel}}{PME}$ hr^{-1}	$\frac{v_{rel}}{v_{uptake}}$	$\frac{V_{max}}{Part.\ P}$ hr^{-1}	$\frac{V_{max}}{chl\ a}$ $nmol\ hr^{-1}$
19 Jul 84	190	--	0.000	0.000	0.0
	180	--	0.000	1.390	137.5
	160	--	0.000	3.526	292.8
	120	--	0.000	23.059	258.8
	60	0.015	0.002	22.969	744.2
8 Aug 84	190	0.003	0.074	0.827	--
	180	0.003	0.005	2.015	269.2
	160	0.008	0.100	2.455	232.9
	120	0.013	0.074	27.746	877.5
	60	--	0.000	34.552	382.5

Table II-10

Significance of Phosphate Release of Phosphate from PME

Clarks Hill Lake

Date	Station	$\frac{v_{rel}}{PME}$ hr^{-1}	$\frac{v_{rel}}{v_{uptake}}$	$\frac{V_{max}}{Part.\ P}$ hr^{-1}	$\frac{V_{max}}{chl\ a}$ $nmol\ hr^{-1}$
19 Jul 84	50	--	0.000	12.000	132.5
	40	--	0.000	7.027	345.5
8 Aug 84	50	0.004	0.004	5.949	157.0
	40	0.004	0.005	0.700	47.3
	30	0.008	0.059	5.537	1,383.4
	20		0.000	7.009	1,027.2

1983). That this increased rate of uptake was a function of living cells, rather than inorganic processes such as siltation, was shown by the sensitivity of phosphate uptake to cyanide poisoning. At most stations in RBR and CHL, greater than 70 percent of the uptake and often greater than 90 percent of the uptake was sensitive to cyanide. The velocity of active uptake per unit amount of particulate P reached very high levels indicating that particulate-P turned over several times per hour. Also, the velocity of active uptake of phosphate per unit amount of "living" chlorophyll a reached very high levels, also characteristic of P-limited algae. Finally, the alkaline phosphatase specific activity (V_{max} of phosphate released from PME in nmol ℓ^{-1} hr^{-1} per nmol particulate-P) reached very high levels, a biochemical response characteristic of planktonic cells growing under P-limitation (Olsen, Knutsen, and Lien 1983).

55. Future studies should emphasize the importance of P nutritional status of the plankton, for a scientific understanding of the functioning of large reservoirs that lead to rational approaches to management. Nutrient limitation also should be documented in traditional ways, using natural phytoplankton in nutrient addition bioassays (Schelske 1984). Especially if P is the only nutrient limiting growth, future studies should focus on those processes that make phosphate available to plankton. Planktonic adaptations to P-limitation should be further studied to document the possibilities for community response to decisions affecting the management of reservoir resources.

56. Many of the results in this study depend on the accurate estimation of phosphate concentration. The calculation of the velocity of phosphate uptake is strictly dependent on phosphate concentration. Other considerations and calculations have in turn been based on the velocity of phosphate uptake. Errors in the estimate of phosphate will be carried to these various determinations. In this study, SRP was used as an estimate of phosphate concentration.

57. A considerable controversy casts doubt on the simple interpretation that soluble reactive phosphorus provides an accurate indication of available orthophosphate. Using primarily a bioassay

procedure, Rigler (1966, 1968) elegantly argued that SRP may often greatly overestimate available phosphate, perhaps by two orders of magnitude. Also using a bioassay, Walton and Lee (1971) demonstrated that laboratory algal cultures, limited in growth only by the availability of phosphate, grew to the extent expected if the molybdenum blue procedure provided an accurate estimate of phosphate concentration. At the other extreme, Cowen and Lee (1976), using a long-term bioassay procedure, have shown that more phosphate than estimated as SRP may be available. Undoubtedly, some of these differences arise from the different types of bioassays used. Those by Lee and his colleagues measured growth due to P availability in the long term; Rigler's radiometric bioassay estimated phosphate availability over the interval of several minutes (i.e., it is an "instantaneous" estimate). Those studies by Lee used laboratory cultures; Rigler's studies followed the rate of phosphate into naturally occurring particles retained on Millipore HA filters. Site-specific and seasonal differences may also affect the interpretation of these discrepant results.

58. Chamberlain and Shapiro (1969) compared SRP and a bioassay with the phosphate detectable after extraction of its molybdyl-derivative into isobutanol. They showed that, in many lakes, the SRP and the phosphate detected after extraction were similar and corresponded to that available to P-starved cultures of Microcystis aeruginosa. In other lakes, the SRP was greater than the extractable phosphate and the biologically available P. In these lakes they showed that the discrepancy between SRP and extractable phosphate was due largely to arsenate in those waters. Arsenate and silicate form molybdyl-derivatives that can be reduced to molybdenum blue, but these molybdyl derivatives are not extracted into isobutanol at pH 8 (Berenblum and Chain 1938, Martin and Doty 1949).

59. Elsewhere, Heath has investigated further the nature of SRP. The chemical and physical properties of SRP were compared with those of phosphate using chromatographic and specific extractive techniques. Filtered lake water was chromatographed by anion exchange chromatography using AG1-X8 (BioRad Labs, Inc.) and the effluent fractions were tested

for SRP by the method of Murphy and Riley (1962). Although not all SRP chromatographed as phosphate, I consistently observed that at least 90 percent chromatographed as phosphate (Francko and Heath 1979). Also in that study, when filtered lake water was chromatographed by Sephadex G-25 gel permeation chromatography, greater than 90 percent of the SRP eluted at the position expected of phosphate. Furthermore, its molybdyl derivative was soluble in isobutanol as phosphomolybdate, demonstrating that SRP, at least for the most part, was not arsenate or silicate (Heath 1986). For these reasons, SRP was used to estimate phosphate concentrations throughout this study.

60. Because of its importance in interpreting the findings regarding results leading to management decisions, the nature of SRP must be carefully determined in RBR and CHL. Specific questions which should be addressed are:

a. Is there a significant interference by arsenate or silicate in determination of SRP?

b. What portion of SRP co-chromatographs with phosphate on Sephadex G-25 and anion exchange resins?

c. What portion of the SRP is really colloidal in size?

d. Does the biochemical uptake of SRP correspond to that expected from chemical studies?

61. Dissolved organic phosphorus compounds could potentially have a great impact on both RBR and CHL. The nutritional significance of DOP depends on the nature of the compounds and the occurrence of those processes necessary to release phosphate from them (Francko and Heath 1979, Heath 1986). This study identified that PME represent substantial portions of DOP detectable on each reservoir. The apparent velocity of phosphate release from these compounds was not sufficiently fast to satisfy the planktonic phosphate demand, but the potential for release of phosphate from DOP is very high. The potential for release is higher than the potential for uptake: the potential PME turnover rate on RBR was greater than 34 hr^{-1} per unit amount of particulate-P, while the potential phosphate turnover was about 6 hr^{-1} per unit amount of particulate-P.

62. If the quantity of PME were to increase in the future, the planktonic community would be capable of using it for growth. This is especially important in RBR, where substantial detrital input from decomposition of submersed vegetation is possible. If large PME detrital inputs to RBR should occur, they would also affect the planktonic community in CHL, which also shows high potential for utilization of PME. Whether increased PME release in RBR would lead to increased PME inputs to CHL or increased phosphate is difficult to predict. In RBR, the potential for release exceeds the potential for uptake--implying that excess outputs of phosphate from RBR may ensue. On the other hand, PME release likely would be gradual and plankton would store the released phosphate intracellularly, attaining a P-sufficient nutritional status--a status that could lead to repression of phosphatase synthesis, which in turn would permit excess quantities of dissolved PME to enter CHL.

63. Future studies should consider this possible impact of detrital decomposition. It is suggested that model in situ studies be conducted to determine the rate at which PME and/or phosphate are released during decomposition. Also, the actual rate of uptake of phosphate from naturally occurring PME should be determined. The velocity of release calculated in this study was presumed to be a reliable estimate of uptake of phosphate released from PME. This calculation provides an indirect observation. Because of its importance in predicting the potential behavior of these reservoir communities, a more direct procedure should be used to show the rate of phosphate release and uptake from PME.

64. RBR and CHL behave similarly in many regards. RBR appears to affect the behavior of the uppermost portion of CHL. In many regards RBR behaves similarly to CHL. In each reservoir the behavior of the planktonic community changed abruptly at the plunge point. The turnover rate of phosphate increased (and the phosphate turnover time decreased) from a slow rate characteristic of P-sufficient communities to a fast rate characteristic of P-limited communities. On each reservoir the uptake of phosphate was dominated by active (cyanide-sensitive)

processes of living cells. On each, the turnover time of particulate-P (velocity of phosphate uptake per unit amount of particulate-P) tended to range from 1 to 10 hr and to decrease toward the dam station. That is, the turnover of particulate-P became progressively more rapid down the length of the reservoir, consistent with the view that the plankton became more nutritionally deficient in P. Also, the velocity of uptake per unit chlorophyll increased greatly toward the dam station--also consistent with the view of a progressively P-limited phytoplanktonic community. This view was supported by a progressive increase in alkaline phosphatase activity and an increasing enzyme specific activity (V_{max} per chl *a*) on each reservoir.

65. Despite these similarities, several notable differences were observed. The input to RBR was high in SRP (especially on 19 July), but decreased abruptly at the plunge point; the input of SRP to CHL from RBR was consistently low, never exceeding 1×10^{-7} mol. The turnover of phosphate in RBR was consistently very slow above the plunge point; the rate of phosphate turnover in CHL was relatively high and was characteristic of P-limited communities on 8 August. Phosphate uptake above the plunge point on RBR was dominated by inactive processes; on CHL, phosphate uptake was dominated by active processes at stations both above and below the plunge point. Although the turnover of particulate-P tended to increase down the length of the CHL reservoir, on 19 July the highest particulate-P turnover on CHL was seen above the plunge point; this was on a day when the turnover on RBR was particularly high. These findings suggest that the input from RBR affected the upper regions of CHL.

66. Future studies should continue to compare these two reservoirs and, if possible, should include Lake Hartwell. By comparing these reservoirs, the generalities in reservoir behavior can be identified. Also, work should continue to determine the ways in which RBR affects the upper regions of CHL and whether these effects become more pronounced over time. Especially, the inputs of PME should be monitored and studied further because of their potential for great impact on the phytoplanktonic community of CHL.

Summary

67. The findings of this study on RBR are consistent with the current view of reservoir function. Incoming water carried an abundant supply of dissolved phosphate. Particles carried into RBR sorb phosphate slowly, and largely through processes not requiring metabolic energy. This incoming water, being cooler and denser, plunges to the bottom of the reservoir as it loses momentum. At this plunge point, it encounters a resident planktonic community in nutrient-poor water. The resident phytoplankton are limited in growth by the availability of phosphorus in the form of phosphate. Uptake of the phosphate that is available is rapid and highly dependent on the metabolic processes of living cells. The biochemical responses of these planktonic cells demonstrated that as flow carries them down the reservoir toward the dam, they become progressively nutritionally starved for phosphorus.

68. Alkaline phosphatase enzymes, produced by these planktonic cells, are capable of releasing phosphate from certain organic compounds dissolved in the water; these compounds are called phosphomonoesters (PME). Phosphatase is produced by cells when growing under P-limiting conditions. Although the amount of phosphatase per cell increased toward the dam station, currently there are insufficient quantities of PME that are processed by phosphatase too slowly to satisfy phytoplankton requirements for phosphorus. However, if detrital decomposition of submersed vegetation in RBR should release large quantities of phosphate or PME into the water, the growth of this P-limited planktonic community would be stimulated.

69. CHL behaves similarly to RBR in many regards. Biochemical evidence shows that its planktonic community also is growth-limited by the availability of phosphate. The biochemical responses of the plankton show a progressive adaptation to P-limitation below the plunge point toward the dam. The water entering CHL from RBR is poor in nutrients. Uptake of phosphate by particles in CHL above the plunge point was done predominantly by metabolically active cells. Phosphatase enzyme activity was seen throughout CHL, showing that PME entering from RBR could

stimulate phytoplankton growth in CHL. These findings indicate that communities in both reservoirs are sensitive to phosphate inputs. Detrital release of phosphate or PME likely would affect both RBR and CHL phytoplankton growth. Management strategies should include a program to control phosphorus inputs. Monitoring of the phosphorus metabolism and nutritional status of the planktonic communities of both reservoirs should be continued.

70. Future work should determine whether P is the only essential nutrient in limiting supply. Also, further studies should be done to establish the dependence on available phosphate, the amount of detectable phosphate that is biologically available, the metabolic responses of the plankton to dissolved organic phosphorus compounds shown to exist in RBR and CHL, and the rate of release of PME from submersed vegetation.

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PART III: PREIMPOUNDMENT AND POSTIMPOUNDMENT ALGAL FLORA OF THE SAVANNAH RIVER AT RICHARD B. RUSSELL RESERVOIR SITE*

Introduction

71. The algal flora of several reservoirs on the Savannah River upstream from Aiken, S. C., have been investigated although the accounts have been limited to post hoc impact-related reports. Interests in the algae have been related primarily to the effects of power plant operations or the management of preexisting reservoirs. However, the Richard B. Russell Reservoir site enjoyed both preimpoundment and postimpoundment investigations. In addition to documenting flora, these investigations provided information on the early development of algal communities in reservoirs.

72. While a number of smaller reservoirs exist and are planned for the upper watershed of the Savannah River drainage basin by private developers, Richard B. Russell Dam and Reservoir completed most of the development potential for this river system. In 1983 this reservoir joined Clarks Hill (completed 1956), Hartwell (completed 1963), Keowee (completed 1971), Jocassee (completed 1973), and numerous smaller reservoirs in converting the major lotic environments to lentic. Located at the head of Clarks Hill Reservoir, Richard B. Russell Dam inundated approximately 48 km of the Savannah River almost to the base of Hartwell Dam (see Figure I-1). The inundation of two tributaries, Rocky River and Beaverdam Creek, also contributed significantly to the planktonic habitat. At power pool, the lake has a surface area of approximately 10,770 ha, an average depth of approximately 11.8 m, and a maximum depth of approximately 53 m. During site preparation, harvestable timber was removed from the lake bed. The remaining trees were left standing except for shoreline clearance. Exposed tops were cut so that at a full

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power pool elevation of 145 m the residue would be submerged at least 3 m below the surface.

Methods

73. Natural assemblages of aufwuchs and macroalgae were collected beginning in 1976 and ending in 1980, well in advance of impact by site preparation. Postimpoundment collections of phytoplankton and aufwuchs began in December 1983. Phytoplankton assemblages were collected monthly as euphotic zone composite samples. Aufwuchs assemblages were collected monthly from both natural (twigs) and artificial (plastic) substrates. Artificial substrates were suspended at the depth of the submerged treetops in order to more faithfully portray the conditions existing among those abundant natural substrates. All collections were preserved with either 3-percent glutaraldehyde or M-3 (Meyer 1971). Diatoms were cleaned by the method of Patrick and Reimer (1966) and mounted using Naphrax. Identifications and counts were made using phase contrast at 500X or brightfield illumination at 1250X magnification.

Results

Preimpoundment flora

74. The completion of Lake Hartwell dramatically altered the physical and chemical characteristics of the Savannah River downstream from Lake Hartwell. Water was released periodically to the Savannah River as a result of the hydroelectric generation schedule at Hartwell Dam. These normal releases were at a rate of approximately 624 cu m/sec or less depending upon the number of turbines in operation. Sediment in the Savannah River below Lake Hartwell was contributed primarily by 210,314 ha of watershed between Hartwell Dam and Richard B. Russell Dam. Most sediments from the upper watershed were trapped by Lake Hartwell and other upstream reservoirs. As a result, streambed substrates for potential colonization varied from bedrock just below Hartwell Dam to sand and silt as well as bedrock near the headwaters of Clarks Hill

Reservoir. The release of hypolimnetic water from Lake Hartwell during the summer stratification period not only moderated water temperatures (annual variation approximately 10° to 20° C) allowing the maintenance of a coldwater fishery, but also ensured availability of inorganic nutrients during times of high insolation. Although the release waters often had low concentrations of dissolved oxygen, the river effectively aerated this water before it reached Clarks Hill Reservoir.

75. The Savannah River presented the botanist with an enormous range of gradients. Daily fluctuations in current, temperature, nutrients, dissolved oxygen, and sediment load commonly occurred throughout the summer and fall. Plant communities also displayed changes along the river, but these changes were often represented only by presence or absence of macrophytes or some other factor influencing substrate quality or availability. From Hartwell Dam to the confluence of Generostee Creek, the aufwuchs was dominated by dense patches of the cyanobacteria, *Lyngbya, Oscillatoria*, and *Phormidium* with associated diatoms and green algae. In general, the number of taxa increased with distance from Hartwell Dam. Between Generostee Creek and the S.C. 181 bridge, the macrophyte *Callitriche heterophylla* was present but not abundant. *Fontinalis filiformis* was also present and its abundance continued on bedrock substrates until the river reached Clarks Hill Reservoir. Between the S.C. 181 bridge and S.C. 184 bridge, substantial growths of *Podostemum ceratophyllum* were observed in the river current on bedrock substrates. *Nitella tenuissima* and *Potamogeton diversifolius* became common on sand and gravel substrates. In particular, *Nitella* formations were observed stabilizing the movement of sand in the streambed. Plant-mediated sand accumulations formed submerged lunate "dunes" with the convex surfaces facing upstream. These formations varied in size from 0.3 m across to large beds in which many of these had merged. *Potamogeton* was usually associated with these beds. Aufwuchs colonized rock surfaces but also accumulated on macrophytes, primarily *Fontinalis* and *Podostemum* which were restricted to the mainstream of the river. *Nitella* and *Potamogeton* were colonized less.

76. Algae of the Rhodophycophyta were observed in the spring and fall in the Savannah River downstream of the S.C. 181 bridge. *Rhodochorton violaceum* and *Batrachospermum boryanum* occurred on rock substrates throughout this portion of the river. *Lemanea australis* was observed in abundance near the S.C. 72 bridge, a few kilometers from Clarks Hill Reservoir.

77. Microscopic algae (Table III-1) were dominated by the Bacillariophyceae. Only a few of the taxa (for example, *Pediastrum* or *Cyclotella*) could be considered phytoplankton released from Lake Hartwell. No distinct trends of distribution were observed downstream from the S.C. 181 bridge. This was due, in part, to the high variability of the substrates and the subsequent variability of the enumerations made from those collections. Cyanobacteria colonized rock substrates more densely than they colonized macrophytes. The other microalgae were evenly represented as both lithophytes and epiphytes.

Post-impoundment flora

78. Development of a phytoplankton community began as early as December 1983, shortly after the lake began to fill. During the months December 1983-April 1984, surface waters were turbid with secchi depths often less than 1 m and relatively cold with temperatures of less than 20° C. During this time, high runoff not only contributed to the turbidity but also influenced the temperature of the lake and the phytoplankton had few dominant species. As a group, the diatoms were relatively more abundant than other groups. Occasional dense populations of flagellates were observed at isolated locations. In February 1984, *Mallomonas teilingii* developed population densities of 180 cells/milliliter at the Beaverdam Creek Station 130 (see Figure I-2) and declined through the spring while *M. caudata* increased to 440 cells/milliliter at Station 130 in March. This trend was repeated at Rocky River Station 140 during April and May 1984. At most stations, however, phytoplankton assemblages began to resemble the assemblages of other reservoirs in the region in May. During May 1984, *Acanthoceras zachariasi* reached a density of 1,400 cells/milliliter at Station 130. Populations of this planktonic diatom increased similarly at other

Table III-1

Algal Flora of the Preimpoundment Savannah River at the Site of Richard B. Russell Reservoir

Cyanochloronta

Anabaena flos-aquae (Lyng.) Breb

Aphanocapsa elachista West & West var. *conferta* West & West

Aphanotheca saxicola Naegeli

Arthrospira jenneri (Kutz.) Stitz

Chroococcus turgidus (Kuetz.) Naegeli

Lyngbya birgei Smith

Microcystis aeruginosa (Kuetz.) Elenkin

Oscillatoria angustissima West & West

O. princeps Vaucher

Phormidium sp.

Chlorophycophyta

Ankistrodesmus falcatus (Corda) Ralfs

Bulbochaete tenuis (Wittr.) Hirn

Closterium ralfsii var. *hybridum* Rabenhorst

Coleochaete nitellarum Jost

Cosmarium botrytis Meneg.

Micrasterias americana (Ehrenb.) Ralfs

Mougeotia sp.

Netrium digitus var. *constrictum* West

Oedogonium kurzii Zeller

Pediastrum duplex Meyen

P. tetras (Ehrenb.) Ralfs

Pithophora oedogonia (Mont.) Wittr.

Rhizoclonium sp.

Scenedesmus quadricauda (Turp.) de Breb.

Staurastrum manfeldtii var. *fluminense* Schumaker

Stigeoclonium subsecundum Kuetz.

Ulothrix zonata (Weber & Mohr) Kutz.

Charophyta

Nitella tenuissima (Desv.) Kutz.

Chrysophycophyta

Bacillariophyceae

Achnanthes exigua Grunow

A. lanceolata Breb.

A. lanceolata var. *dubia* Grunow

A. levanderi Hust.

A. linearis f. curta H. L. Smith

A. linearis var. *pusilla* Grunow

A. microcephala Kutzing

A. minutissima Kutzing

Anomoeoneis serians var. *brachysira* (Breb.) Hustedt

A. vitrea (Grunow) Ross

Asterionella formosa Hassall

(Continued)

Table III-1 (Continued)

Cocconeis placentula (Ehrenberg)

C. placentula var. *lineata* (Ehrenb.) Cleve

Cyclotella menghiniana Kutzing

C. stelligera Cleve & Grunow

Cymbella laevis Naegeli

C. lunata W. Smith

C. microcephala Grunow

C. microcephala var. *crassa* Reimer

C. minuta Hilse ex Rabenhorst

C. minuta var. *silesiaca* (Bleisch ex Rabh.) Reimer

C. triangulum (Ehrenberg) Cleve

C. tumbia (Breb.) van Heurck

Denticula tenuis Kutzing

Diploneis puella (Schumann) Cleve

Eunotia curvata (Kutz.) Lagerst.

E. curvata var. *capitata* (Grunow) Woodhead & Tweed

E. formica Ehrenberg

E. naegelii Migula

E. quaternaria Ehrenberg

E. vanheurckii Patrick

E. zasuminensis (Cabejsz.) Korner

Fragilaria construens var. *binodis* (Ehrenb.) Grunow

F. crotonensis Kitton

F. vaucheriae (Kutzing) Peters

Frustulia assymetrica (Cleve) Hust.

F. rhomboides (Ehrenberg) de Toni

F. rhomboides var. capitata (A. Mayer) Patrick

F. rhomboides var. *crassinervia* (Breb. ex W. Smith) Ross

F. rhomboides var. *saxonica* (Rabh.) de Toni

F. vulgaris Thwaites

Gomphonema acuminatum Ehrenberg

G. gracile Ehrenberg

G. helveticum Brun.

G. subclavatum var. *mexicanum* (Grunow) Patrick

G. parvulum Kutzing

G. subtile Ehrenberg

G. tenellum Kutzing

G. truncatum var. *capitatum* (Ehrenberg) Patrick

G. truncatum var. *turgidum* (Ehrenberg) Patrick

G. ventricosum Gregory

Gyrosigma nodiferum (Grun.) G. West

Hantzschia amphioxys (Ehrenberg) Grunow

Melosira distans (Ehrenb.) Kutzing

M. granulata (Ehrenberg) Ralfs

M. varians C. A. Agardh

Meridion circulare Agardh

Navicula capitata Ehrenberg

N. caroliniana Patrick

(Continued)

(Sheet 2 of 3)

Table III-1 (Concluded)

N. elginensis (Gregory) Ralfs
N. exigua (Gregory) O. Muller
N. halophila (Grunow) Cleve
N. hambergii Hustedt
N. jaernfeltii Hustedt
N. minima Grunow
N. mobiliensis Boyer
N. mutica Kutzing
N. mutica var. *tropica* Hustedt
N. notha Wallace
N. radiosa Kutzing
N. radiosa var. *parva* Wallace
N. simula Patrick
Neidium affine (Ehrenberg) Cleve
Nitzschia acicularis W. Smith
N. amphibia Grunow
N. clausii Hantzsch
N. dissipata (Kutzing) Grunow
N. fonticola Grunow
N. gracilis Hantzsch
N. ignorata Krasske
N. lorenziana Grunow
N. palea (Kutzing) W. Smith
N. parvula Levis
Rhizosolenia eriensis H. L. Smith
Stauroneis phoenicenteron Ehrenb.
Surirella angustata Kutzing
S. linearis W. Smith
S. robusta Ehrenberg
Synedra acus Kutzing
S. famelica Kutzing
S. planktonica Hains & Sebring
S. pulchella Kutzing
S. rumpens var. *meneghiniana* Grun.
S. socia Wallace
S. ulna (Nitzsch) Ehrenberg
S. ulna var. *oxyrhynchus* Kutzing
Tabellaria fenestrata (Lyngb) Kutzing
T. flocculosa (Roth) Kutzing

Rhodophycophyta

Batrachospermum boryanum Sirodot
Lemanea australis Atkinson
Rhodochorton violaceum (Kutz.) Drew

(Sheet 3 of 3)

locations during June 1984. Likewise *Rhizosolenia eriensis*, another planktonic diatom developed densities of 560 cells/milliliter at Station 130 during May and 1,200 cells/milliliter in June. During June, *R. eriensis* also increased population densities at other reservoir locations. In addition, other planktonic algae showed similar trends of increase during May and June 1984.

79. Nutrient availability alone explains much of the apparent tendency of the phytoplankton to develop first in the tributary arms of the reservoir, since highest nutrient concentrations were observed here (see James et al. 1985). At most locations, surface water remained low in dissolved nutrients. Soluble reactive phosphorus and nitrate concentrations were often unmeasureably low or in the microgram per liter range (James et al. 1985). However, water movements during filling may have influenced phytoplankton development as well as other lake processes. The tributary arms were undoubtedly quiescent relative to water movements in the main channel although actual measurements of such movements are lacking.

80. Although the postimpoundment phytoplankton assemblage (Table III-2) was dominated numerically by diatoms, Chlorophycophyta were also important and, with the diatoms, typify phytoplankton assemblages of Richard B. Russell and many other southeastern reservoirs and lakes. The widespread genera, *Ankistrodesmus, Scenedesmus, Selenastrum, Staurastrum, Pediastrum*, and *Crucigenia* as well as *Dinobryon, Rhizosolenia, Cyclotella, Asterionella, Fragillaria, Melosira, Nitzschia*, and *Synedra* are common in other reservoirs in the Savannah River drainage as well as in Richard B. Russell. Most of these genera were not common in the aufwuchs of the preimpoundment Savannah River. Those genera that were common to both aufwuchs and phytoplankton, such as *Nitzchia* or *Synedra*, were represented by different species. For example, *Nitzschia palea*, a common aufwuchs diatom, is unlikely to be found in phytoplankton although the opposite is true for *N. acicularis*. Likewise, *Synedra ulna* is typical of the aufwuchs while *S. planktonica* is euplanktonic.

81. Aufwuchs in Richard B. Russell Reservoir developed initially dense accumulations on the newly submerged treetops during January and

February 1984. These accumulations were dominated by the filamentous green algae *Oedogonium* and *Mougeotia* and the diatoms *Tabellaria*, *Synedra*, and *Gomphonema*. The nannandrous *Oedogonium braunii* was commonly observed with sexual reproductive structures while *Mougeotia* was never observed in sexual reproduction. In addition, *Gomphonema* was commonly observed forming gelatinous envelopes containing two to four cells lying side by side. Despite their similarity to auxospores (Smith 1950), these were more likely unusual gelatinous vegetative formations by diatoms.

82. As the lake filled and natural substrates were submerged to greater depths, the aufwuchs became sparse then absent. By May 1984 the dense growths observed in February had declined and, by late June, both natural and artificial substrates submerged to depths of 3 m or greater were barren of algae. At least 10 percent of incident light was available at those depths (based on Secchi disc transparency); however, the distribution of aufwuchs seems to have been more strongly associated with the availability of dissolved oxygen. The inundation and subsequent decomposition of a large mass of organic material allowed a large biological oxygen demand. A well-defined thermal discontinuity existed at or near 4-m of depth at most locations, and dissolved oxygen concentrations commonly declined dramatically at this thermocline. Objects at depths shallower than 4 m accumulated aufwuchs similarly to other reservoirs, and substrates at greater depths did not.

83. Clearly, the preimpoundment flora contributed little to the postimpoundment flora. Phytoplankton taxa probably had their origin in Lake Hartwell or Lake Secession (located on Rocky River). The long-term development of this reservoir must still consider the aufwuchs as a potentially important source of algal production because conditions which initially limited algal growth may subside as the more labile organic mass (leaf litter, etc.) decomposes. Surprisingly, the rich flora of the preimpoundment aufwuchs likewise contributed little to the postimpoundment flora. Initial colonization by common cosmopolitan species but not by taxa such as *Synedra pulchella* or numerous *Navicula* species may have been due to rapid depletion of oxygen or a related

Table III-2

Postimpoundment Algal Flora of Richard B. Russell Reservoir

Cyanochloronta

Anabaena flos-aquae (Lyng.) Breb.

Lyngbya birgei Smith

Merisompedia elegans A. Br.

Oscillatoria geminata Meneghini

Chlorophycophyta

Ankistrodemus falcatus (Corda) Ralfs

Arthrodesmus incus (Breb.) Hassall

Bulbochaete tenuis (Wittr.) Hirn

Chlamydomonas sp.

Chodatella subsalsa Lemmermann

Closterium ralfsii var. *hybridum* Rabenhorst

Cosmarium botrytis Meneg.

C. regnesi var. *montanum* Schmidle

Crucigenia crucifera (Wolle) Collins

Dictyosphaerium planctonicum Tiffany & Ahlstrom

Gloeocystis vesiculosa Nageli

Golenkinia radiata Chodata

Micratinium pusillum var. *longisetum* Tiffany & Ahlstrom

Mougeotia sp.

Oedogonium braunii Kutzing

Oocystis borgei Snow

Pediastrum duplex Meyen

P. tetras (Ehrenb.) Ralfs

Quadrigula lacustris (Chodat) G. M. Smith

Scenedesmus abundans var. *assymetrica* (Schroed.) G. M. Smith

S. bijuga (Turp.) Lagerheim

S. braziliensis Bohlin

S. obliquus (Turp.) Kutzing

S. quadricauda (Turp.) de Breb.

Selenastrum westii G. M. Smith

Sphaerocystis schroeteri Chodat

Spondylomorum quaternarium Ehrenb.

Staurastrum manfeldtii var. *fluminense* Schumaker

Stigeoclonium subsecundum Kuetz.

Tetraedron trigonum (Nag.) Hansgirg

Ulothrix zonata (Weber & Mohr) Kutz.

Euglenophycopyta

Phascus sp.

Rhabdomonas costata (Korshik) E. G. Pringsheim

Trachelomonas hispida (Perty) Stein

Cryptophycophyta

Cryptomonas erosa Ehrenberg

Chrysophycophyta

(Continued)

Table III-2 (Concluded)

Xanthophyceae

Pseudotetraedron neglectum Pascher

Chrysophyceae

Chromulina nebulosa Pascher

Dinobryon sertularia Ehrenberg

Mallomonas caudata Conrad

M. teilingii Conrad

Ochromonas mutabilis Klebs.

Chloromonadophyceae

Gonyostomum semen (Ehrenb.) Diesing

Bacillariophyceae

Asterionella formosa Hassall

A. ralfsii W. Smith

Acanthoceras zachariasi J. Brun

Cyclotella meneghiniana Kutzing

C. pseudostelligera Hustedt

C. stelligera Cleve & Grunow

Eunotia zasuminensis (Casbesz.) Korner

Fragilaria crotonensis Kitton

Gomphonema acuminatum Ehrenberg

G. gracile Ehrenberg

G. parvulum Kutzing

Melosira distans (Ehrenb.) Kutzing

M. granulata (Ehrenberg) Ralfs

M. varians C. A. Agardh

Navicula notha Wallace

N. radiosa Kutzing

N. radiosa var. *parva* Wallace

Nitzschia acicularis W. Smith

N. lorenziana Grunow

N. palea (Kutzing) W. Smith

Rhizosolenia eriensis H. L. Smith

Synedra acus Kutzing

S. famelica Kutzing

S. planktonica Hains & Sebring

S. rumpens var. *meneghiniana* Grun.

S. ulna (Nitzsch) Ehrenberg

S. ulna var. *oxyrhynchus* Kutzing

Tabellaria fenestrata (Lyngb) Kutzing

T. flocculosa (Roth) Kutzing

Pyrrhophycophyta

Ceratium carolinianum (Bailey) Jorgeson

Peridinium cinctum (Muller) Ehrenberg

chemical limitation. However, the composition of the aufwuchs in nearby reservoirs is similar, which suggests that the taxonomic composition of the postimpoundment aufwuchs is never likely to resemble the preimpoundment aufwuchs.

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PART IV: THE SPATIAL DISTRIBUTION OF ZOOPLANKTON IN RICHARD B. RUSSELL LAKE*

Introduction

84. The relationship between zooplankton distribution and the physical and chemical state of lakes has been the focus of a number of studies (Patalas 1971, Spurles 1984). Data concerning zooplankton distribution and diversity have been used as an indicator of lake physical and chemical conditions (Pennak 1957, Woodell 1970) and as a manifestation of lake succession (Moss 1980). As such, the characterization of zooplankton distribution can provide valuable information concerning the biological state of an aquatic ecosystem.

85. Richard B. Russell reservoir affords the opportunity for the analysis of changes in zooplankton diversity and distribution in a newly impounded reservoir. The integration of physical and chemical data with information on community composition and functional relationships allows the assessment of the effects of environmental fluctuations on the biota in this system. The information gained from this study should provide fundamental insights into the process of zooplankton colonization and community stabilization during an early successional stage.

Materials and Methods

86. The zooplankton were investigated with respect to vertical and longitudinal distribution. General vertical distribution was determined by taking hypolimnetic and epilimnetic net tows at seven sampling sites. The seven sampling sites were chosen to provide information on the distribution of the zooplankton.

87. Change in community structure was considered to be an indication of successional change within the reservoir (Ricklefs 1973). The

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sum difference successional index (Lewis 1978) was calculated for the total number of organisms over all sites on each date. This index allowed assessment of changes in species abundance, at a particular date, relative to all constituent species abundances in the community.

88. In addition to the specific characterization of the community succession, each of the recorded organisms was placed into one of three functional groups based on known feeding strategies according to the criteria of Le Cren and Lowe-McConnell (1980). Spurles (1979) feels that functional groupings according to feeding ecology may be better indicators of the effects of environmental conditions on a zooplankton community than customary taxonomic classification. In addition, the large number of species and individuals obscured determinations of patterns or changes in community structure. The distribution of the zooplankton should be a reflection of their physiological requirements relative to their environment (Parsons 1980). Assuming that species within a functional group have somewhat similar physiological demands, the dominant functional group should reflect the general state of their immediate surroundings.

89. The processes of zooplankton colonization and community stabilization were tracked by observing changes in the percent functional group per total community. The percent functional group values allowed comparison of abundances on a given date, at a given site. Analysis of the changes in the sum difference successional index and percent functional group revealed both successional changes over the entire lake and the community structure at particular sites and on particular dates.

90. Approximately every 2 weeks, epilimnetic and hypolimnetic tows, using an 80-mm-mesh, 0.5-m Birge closing net, were taken at Stations 80, 100, 120, 130, 140, 160 and 180 in Richard B. Russell Reservoir (see Figure I-2). Tows were taken while anchored to the buoy marking the sampling site. The zooplankton were concentrated in the net bucket by rinsing the exterior of the net with lake water. The bucket was rinsed and drained into a 125-ml plastic bottle. The samples were preserved in the field with 95-percent ethyl alcohol and Lugol's iodine (Lind 1979). The bottles were labeled and returned to the laboratory.

After settling, the 125-ml samples were concentrated to 10 ml, and three 1-ml alioquots were removed from each thoroughly mixed sample. Each alioquot was placed in a Sedgewick-Rafter cell and covered with a slip, and the zooplankton were taxonomically classified and enumerated with the aid of a compound microscope. Thirteen species of zooplankton, one juvenile insect species, and two juvenile zooplankters were cataloged over the sampling period, which lasted from May 21 to October 18, 1984. Each of the recorded organisms was placed into one of the following functional groups and known feeding strategies according to the criteria of Le Cren and Lowe-McConnell (1980):

Fine Particle Feeders (FPF)

1. Rotifers-*Keratella, Kellicota*
2. *Bosmina longirostris* (OFM)
3. Juvenile Cladocerns
4. Ostracods
5. Chydorids
6. *Ceriodaphnia reticulata*
7. Copepod naupli

Large Particle Feeders (LPF)

1. *Daphnia parvula* Fordyce
2. *Diaphanasoma brachyurum*
3. *Holopedium amazonicum*
4. *Daphnia pulex* Leydig

Raptors

1. *Cyclops scutifer* Sars
2. *Ectocyclops phaleratus*
3. *Leptodora kindti* Focke
4. Chaoborinae
5. Diaptomus species

91. The change in the percent functional group per total community was then determined according to the following formula:

$$\text{Percent of functional group} = \frac{\text{Total number of species } i}{\text{Sum of individuals over all species } i \text{ to } x}$$

92. The community successional index was also calculated for the total number of organisms over all sites per date according to the formula of Lewis (1978).

Sum difference successional index =

$$s = \frac{\text{sum of } i\ [b_i(t_1)/B(t_1)] - [b_i(t_2)/B(t_2)]}{t_2 - t_1}$$

where:

$b_i(t)$ = abundance of the i^{th} species at time t

$B(t)$ = the size of the community at time t

93. This index allows the assessment of changes in species abundance b at a date t relative to all the constituent species abundances B in the community. Large successional indices indicate large changes in relative species abundances, a condition that is indicative of successional change.

Results

94. While minimal information was obtained about vertical distribution, it was found that hypolimnetic and epilimnetic tows differed in abundances and diversities of species. During periods of stratification, organisms were found in both the epilimnion and hypolimnion, but the greater abundances and diversities were found in the epilimnion. Rotifers were the most abundant of the species found in the hypolimnion.

95. Relative to the longitudinal distribution, FPF were the most abundant and ubiquitous group. On May 21, 1984, the raptors were the most pronounced group at every site, excluding Station 140. All the sites, except Station 140, were approximately isothermal and mixed at this time. Station 140, which was stratified, was dominated by FPF. On May 21, approximately 253 organisms per liter, the total for all sites, were counted. Between May 21 and 31, the rate of succession was low (0.0391), indicating little change in relative species composition at the time. On May 31 the FPF represented the greatest percent of the total abundance at every site excluding Stations 130, 180, and 160. Stations 180 and 160 were not stratified and still had high percentages

of raptors. While Station 130 was stratified at that time, it still had a 50-percent dominance of raptors. The total number of organisms for the sampling date was 214 per liter, a number that was approximately the same as the May 21 total count. However, the successional rate calculated for the period May 31-June 13 was 0.0637, a number that is indicative of a community change. This larger successional index corresponds to the change from raptor dominance on May 21 to the FPF dominance that was observed on May 31.

96. The FPF dominated all sites except Station 140 by June 27; however, the total number of organisms sampled was 170 per liter, the lowest observed on any sampling day. The successional index also dropped to 0.0260 between June 13 and June 27. However, from June 27 through July 25, the succession rate reached 0.0500, indicating a significant shift in the relative species abundances. This corresponds to the continued increased in FPF dominance. During this time there was also a marked increase in the total number of animals/per liter. On July 11 approximately 1,366 organisms per liter were counted, and similar estimates persisted through early August.

97. From July 11 to October 18, the FPF were the dominant functional group at every site and the rotifer constituent was the major contributor to the FPF numbers. Rotifers numbers reached 1,140 animals per liter. The next highest count, 100 animals per liter, was recorded for *Bosmina logirostris*, another FPF. The successional index decreased to 0.0129 for the period September 6-18, indicating little change in relative species abundances. Between September 18 and October 2 the successional index increased from 0.0129 to 0.0396. However, the FPF maintained a functional dominant (90 percent or greater) at every site, and there was no major change in the overall abundance of zooplankton. Between October 2 and October 18 the successional rate index fell to 0.0070, with a stable dominance of FPF throughout the lake.

98. While FPF dominated the functional group categories of the zooplankton in the reservoir, the LPF and raptors generally remained numerically stable. The LPF showed a slight increasing trend until August 6, when the numbers peaked at an average of 78.99 organisms per

liter. The raptor numbers began high and then dropped to 70 ±30 organisms per liter. Neither raptors nor LPF attained the abundances of the FPF. Following stratification, the LPF were most abundant at sites other than Stations 130 and 140. The raptors, however, were found in the arms as well as the main basin of the lake with equal regularity.

99. The sites with the greatest total numbers of zooplankton were Stations 130 and 140. Station 130, Beaverdam Creek, had the greatest number of organisms per liter on 6 of the 12 sampling dates. Rocky River, Station 140, had the highest abundances on 3 of the 12 dates sampled. Additionally, Station 180, the site located nearest Hartwell Dam, had the least number of organisms per liter throughout the sampling period.

Discussion

100. The results indicate that stratification affects the abundance and distribution of zooplankton species and functional groups in this reservoir. While the most abundant and diverse samples were taken from the epilimnion, rotifers, members of the FPF, were the most prevalent organisms in the hypolimnion.

101. Longitudinal distribution was also affected by stratification. Prior to stratification, all sites were dominated by the raptors, albeit they and the other functional groups appeared in moderate numbers. At the same time, the successional rate index was relatively low, indicating little successional change. With the onset of stratification the successional rate index increased and the FPF increased in numbers relative to the LPF and raptors. The total number of organisms per sampling date also increased significantly during this period. The rotifer constituent of the FPF was the outstanding cause for this increase in total numbers and the dominance of the FPF. During this period the LPF and raptor counts had not changed significantly, indicating that the FPF were not only capable of tolerating stratification conditions but were also able to exploit the environment by increasing their numbers beyond those of LPF and raptors. While the LPF and

raptors do not appear to be adversely affected by the increase in FPF or stratification, some other parameter may be restricting their population size. Both LPF and raptors were found throughout the reservoir, but Stations 130 and 140 did not attain the LPF abundances that were seen in the main basin. Combined, Stations 130 and 140 had the greatest total number of organisms on 9 of the 12 dates sampled. Site conditions appeared to restrict the LPF and not the FPF. Stations 130 and 140 were located in the two arms of the reservoir and often had shallower epilimnia than the other sites. Perhaps the LPF and raptors, which possibly require greater oxygen concentrations than FPF, are spatially restricted during periods of stratification.

102. Food resources would not be expected to limit any of the functional groups. FPF rely on bacteria and detritus for food, and it is reasonable to assume that a newly impounded reservoir should have adequate food resources. LPF are live-algae feeders and will feed on small rotifers and protozoans; bacteria and detritus are not primary nutritional resources. Assessment of algal counts (See Part III) revealed a number of large algal types; rotifers were also plentiful. The main food resources for raptors are small particle feeders, and there appeared to be sufficient resource supplied in the large numbers of FPF. Food resources apparantly were not limiting the raptor or LPF population sizes, although predation may have. The relaxed predation on the FPF may account for the increase in numbers. However, the relationship between the increase in FPF and the onset of stratification indicated that environmental conditions were the primary controls on community composition. If the community structure was controlled by predation, the number of FPF should have increased prior to stratification. This was not the case. Hence, differing physiological tolerances among the groups to stratification may have constrained the LPF and raptor populations while favoring the FPF. Further support of this assertion would be found if fall mixes were accompanied by an increase in the successional rate index and a change in the dominant species to LPF or raptors.

Summary/Conclusions

103. Stratification of Richard B. Russell reservoir affects the vertical and longitudinal distribution and abundance of zooplankton. Epilimnetic and hypolimnetic tows revealed a distinct vertical distribution at all sites. The most diverse and abundant samples were epilimnetic. Of the zooplankton seen in the hypolimnetic samples, rotifers (a member of the FPF) were most common. Prior to stratification, all seven sites were dominated by raptors. The onset of stratification was accompanied by a change in dominance to FPF. FPF appeared to be physiologically tolerant of stratification conditions and able to increase in numbers when the LPF and raptors could not. This "physiological control" hypothesis for the zooplankton community composition can be tested by further sampling and analysis to show a change in the dominant functional groups in the reservoir with the onset of fall mixes.

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PART V: MEASUREMENT OF INUNDATED FOREST BIOMASS AT RICHARD B. RUSSELL LAKE*

Introduction

104. The filling of Richard B. Russell Lake resulted in the inundation of approximately 26,000 acres of land along the Savannah River and its tributaries. Although selected areas of the basin and all areas above 462 ft NGVD were clear-cut, nearly 9,000 acres of forested land were inundated during filling. This material, therefore, represents a significant potential source of organic material and provides structure for colonization by organisms. Because of its potential importance to water and environmental quality, attempts were made to quantify this material.

Methods and Results

Phase I

105. This phase involved the collection of 203 samples using Basal Area Factor (BAF) 10-points distributed throughout the various types of vegetative areas flooded by Russell Lake (Table V-1). Several basic steps were followed in locating and measuring these ground samples. The area to be included in the study (forested land covered by Russell Lake, henceforth called the study area) was first outlined on clear acetate overlays of the Corps of Engineer color aerial photographs of Russell Lake taken "after clearing" but before the level of the lake was raised. Only forested areas were included in this delineation. A few areas had been cleared so thoroughly that only brush, vines, and an occasional small tree were present. Such areas were not included in the study area, nor were islands with vegetation standing above the level of lake.

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* Part V was written by William A. Shain, Consulting Forester, Clemson, S. C.

Table V-1

Distribution of the BAF 10 Ground Samples in the Various Vegetative Types of RBR Lake, 1984

Vegetative Type/Density	Number of Ground Samples
Bottomland hardwood	
Low	19
Medium	25
High	26
Total	70
Upland hardwood	
Low	26
Medium	24
High	7
Total	57
Pine hardwood	
Low	11
Medium	8
High	4
Total	23
Bluff hardwood	
Medium	10
High	10
Total	20
Cutover - low dens ty	25
Hardwood - edge of river	9
Total - all types	203

106. Once the forested land had been outlined on the aerial photographs, a short preliminary set of land areas was identified on the photos to check the classification scheme and to become familiar with the various vegetative types in the Russell Lake basin. (Descriptions of the vegetative types are given below.)

107. Next, a detailed examination was made of the photos to determine areas adjacent to the flooded timberland that would be representative of the forested area covered by Russell Lake. The four sections of the lake (the main basin, the Rocky River section, the Savannah River section, and the Beaverdam section) were examined closely to determine appropriate areas in which to locate the ground samples. Care was taken to locate a large proportion of ground samples on river islands and on bottomlands adjacent to the major drainages. This meant that the clusters of three ground samples were located mainly in the upper stretches of the Rocky River and the Savannah River.

108. Because the ground samples had to represent the basic vegetative types already covered with water, the ground samples had to be specifically located in vegetative types similar to the inundated areas. Thus, this sample of ground points was not a random, unbiased sample. Each cluster of three points was selected using the aerial photographs to locate a particular type. For instance, after photo-location of a suitable stand, the field crew would leave a particular tree on the edge of the lake and go north one chain to the first point. The second point would then be two chains north of point one and the third, two chains north of point 2. If the sample area was exactly like the type needed, another cluster of three points was located in the same general area.

109. While the ground samples were somewhat biased in their location, we are confident that they are representative of the specific vegetation types flooded by the impoundment. Selecting ground samples in this manner will provide the most accurate assessment of the green tons per acre of biomass under RBR Lake that was possible under the conditions of this study.

110. The following vegetative types were identified during Phase I:

a. Bottomland hardwood. This type includes sites adjacent to rivers and streams and is normally characterized by being located in the floodplain of a drainage. The soil types are generally fertile loamy sands and sandy loams. Tree species include water and willow oaks, soft hardwoods such as yellow poplar and sweetgum, and such mesophytic trees as river birch, sycamore, and black willow. Each bottomland hardwood site was also classified as low, medium, or high density.

b. Upland hardwood. This type includes those sites above the level of the more fertile bottomlands, generally located on the middle slopes, upper slopes, and ridges adjacent to the floodplain of the drainage. Soils are predominantly of the red clay types. Some common tree species were red oaks, white oaks, sourwood, hickory, and redcedar. Each upland hardwood site was classified as low, medium, or high density, although high-density upland hardwoods were rare in this study area because such areas have been previously logged.

c. Pine hardwood. Preliminary examination of the aerial photos indicated that very little pine timber was left standing to be flooded by the lake. Still, several areas had a significant proportion of pine timber. Also, there were scattered pines in nearly all the types, even those that on aerial photographs appeared to be pure hardwoods. Ground samples were selected in such a manner that some pines were naturally included so as to represent the pine portion of the study area properly. Practically all of the pine hardwood types were low- and medium-density classes.

d. Bluff hardwood. Because there are natural bluffs along many sections of the Savannah River and a few sections of its tributaries, a category was included to delineate this type. These areas are extremely steep, rocky, 100 to 200 m wide, and have a preponderance of very large, cull hardwoods. Only medium- and high-density classes are included in this type because a bluff site that would be classified as low density would normally be included in the low-density upland hardwood category.

e. Edge of river hardwoods. Many areas immediately adjacent to the Savannah River and its tributaries have large-crowned and often cull hardwoods leaning out over the water. Because these areas were probably not economically feasible to log when the lake basin was cleared, this class was included. The main species are river birch, sycamore, water and willow oak, and yellow poplar. Only one class of combined medium- and high-density timber was included for this type.

f. Cutover - low density. A portion of the study area was logged and all commercial timber salvaged before the RBR Lake began filling. Special care was taken to include samples that matched these cutover areas. Many of the ground samples representing the cutover type were located on islands scattered throughout the lake.

111. Each type was normally categorized as low, medium, or high density depending upon the basal area of the stand.

a. Low density - normally less than 40 sq ft of basal area per acre. Care was taken to also incorporate the volume of the ground sample into the density classification. For example, a stand that had about 40 sq ft of basal area but with short, small-volume trees, was classified as low density while a stand with the same basal area but with tall, high-volume trees was classified as medium density. Therefore, site quality (height of trees) was included as a factor influencing stand biomass.

b. Medium density - normally from 40 to 90 sq ft of basal area per acre.

c. High density - normally over 90 sq ft of basal area per acre.

112. The basic field sample design was formulated to select a series of stands for sampling that were similar to those in the study area. Within these selected stands, a cluster of three BAF 10-points was taken, the first point of which was normally located one chain from the edge of the lake or a road directly into the stand. Each of the subsequent points was located two chains from the previous one and in a cardinal direction from it. Some deviation from this selection system was necessary in those stands that were narrow in width (e.g., the narrow bands of bluff hardwoods and the hardwoods along the edge of streams and rivers). In these cases, an attempt was made to keep the center of ground samples in the middle portion of the selected stand.

113. At each selected site, the center of the point was cleared of debris and a piece of flagging was tied to the nearest tree. All trees around this point center were checked to see if they would be sampled with a BAF 10 prism. All selected trees were first marked with a spot of paint and tallied by species, diameter at breast height (dbh), merchantable height, and total height. All dbh measurements were measured

to the nearest 0.1 in., and height measurements were estimated to the nearest 5 ft with frequent checks for accuracy. A tree was designated as cull if it was over 50-percent defective. Trees 4.6 in. dbh and larger were tallied by dbh, merchantable height, and total height. Trees 3.6 to 4.5 in. dbh were tallied by dbh and total height and marked nonmerchantable. A 1/100-acre plot was taken up and down the slope in a 1/100-acre rectangular plot 6 x 72.6 ft. On this rectangular plot, saplings were tallied by dbh and total height and classed as understory.

Phase II

114. Phase II involved delineation of aerial photographs (approximately 100) by timber type, measurement of areas of each type, assessment of timber total heights, and determination of crown closure for each type. Aerial photos covering the study area were identified and examined in the third dimension to ensure that only those portions of timber to be covered by water would be included in the evaluation. Those portions of forested land were outlined on clear acetate overlays for each photograph. Care was taken to utilize the central portion of each photograph so that a minimum amount of displacement would be apparent. Only those areas that had measurable amounts of forest biomass were included in the study area. Excluded were areas covered with brush, vines, and an occasional small tree.

115. Once the forested portions of the study area were delineated on the aerial photographs, these areas were further divided into vegetative types to include bottomland hardwoods, upland hardwoods, pine hardwoods, bluff hardwoods, edge of river hardwoods, and low-density cutover areas. A summarization of the acreages in each of these types is presented as Table V-2.

116. Crown closure was evaluated to present the three density classes utilized in the 203 ground samples. Low density generally corresponds to 0 to 30 percent crown closure; medium density, to 31 to 69 percent crown closure; and high density, to 70 to 100 percent crown closure. Some exceptions were made in areas where it was apparent that there was a large population of large trees. Such areas were classified

in the higher category on many occasions in a manner similar to the classification of density in the ground samples.

117. Acreages in each vegetative type were measured by using a dot grid to assess the number of dots in each type on each photograph. The dot grid used in this study had 64 dots per square inch, and each dot represented approximately 0.35 acre. In converting the dot grid count from each type to the area in acres, the exact scale of each photograph was calculated first by comparing a measured photo distance to the same distance on a US Geological Survey 7.5-min. quad map.

118. Timber total heights were accounted for by using the actual measured tree heights for each ground sample tree as a base from which to correlate the data from 203 ground samples (approximately 1,600 sample trees) to the vegetative types on the aerial photographs.

Phase III

119. Green biomass weights were calculated in Phase III of the study. Green biomass was defined as the undried weight of wood and bark for aboveground portions of the trees; foliage was included for overstory pines but not for hardwoods. These calculations were aided by the computer program CALBIO, which was developed by the Department of Forestry at Clemson University. The equations used in this program are presented in Table V-3 and are based on sources of information provided in Table V-4. The equations predict the biomass of the various classes of forestry products described in Table V-5.

120. Calculation of green biomass weights on a unit-area basis for each vegetative type and density class were based on information for the 203 BAF samples. These data are presented in Table V-6. Total green weight biomass for flooded forest areas in each of the major basins was then determined based on measured areas (Table V-2) and biomass per unit area (Table V-6). These data are presented in Table V-7.

121. Of the 8,642 acres of flooded forest land under Russell Lake, over 1,221 acres (14 percent) were left in an untopped condition for the purpose of improving fish habitat. There are 13 major fish tree areas, located in various parts of the lake. These areas represent nearly 16 percent of the total forest biomass in the lake basin. Because these

flooded stands are situated in relatively shallow water near shore, a substantial portion of their biomass is presently located above the full pool lake surface in tree crowns and upper main stems. This above-surface biomass will eventually break down from decay, wind, and wave action, and become submerged or float to shore. This being the case, the total given for flooded forest biomass in Table V-7 is an overestimate because a part of this biomass is not presently (1984) under water.

122. To correct the total lake estimate, individual estimates for total biomass in each of the 13 major fish tree areas were broken down into above- and below-surface components (Table V-8). These methods are discussed in Appendix A. The above-surface portions from each area were summed, and this resulting figure was subtracted from the total lake biomass estimate (Table V-9). In this way, the total submerged forest biomass in Russell Lake was estimated.

Table V-2

Area of Flooded Forest (in Acres) in Russell Lake, as Determined from Measurements of 1982 Color Aerial Photographs

Flooded Area	Vegetative Type/Density												
	Bottomland Hardwood			Upland Hardwood			Pine-Hardwood			Cutover (Low)	Bluff* Hardwoods	Hardwood* River Edge	Total
	Low	Medium	High	Low	Medium	High	Low	Medium	High				
Beaverdam Creek	34.15	34.32	25.80	187.99	668.43	59.89	--	129.12	[illegible]	[illegible]	--	[illegible]	[illegible]
Lower Main Lake Basin**	348.86	953.05	114.64	127.07	206.63	34.87	--	85.06	--	[illegible]	[illegible]	[illegible]	[illegible]
Rocky River	254.69	864.57	95.37	47.02	176.14	16.73	--	--	--	[illegible]	[illegible]	[illegible]	[illegible]
Savannah River	636.92	2,162.67	211.71	147.50	646.62	17.38	1.78	85.47	[illegible]	[illegible]	[illegible]	[illegible]	[illegible]
Total flooded forest area	1,274.62	4,014.61	447.52	509.58	1,697.82	128.87	1.78	299.65	[illegible]	[illegible]	[illegible]	[illegible]	[illegible]

* Only one crown density class.

** Downstream from confluence of Beaverdam Creek.

Table V-3

Green Weight* Biomass Prediction Equations

Section A. Overstory Trees

1. Soft Hardwoods

Total tree = $0.12113\ D^2H^{1.02190}$

Pulpwood portion of main stem (stump to 4-in. diameter inside bark (dib)) = $\frac{0.02950\ D^2H^{1.14599}}{0.06869\ D^2H^{1.08516}}$ · Total Tree

Sawlog portion of main stem (stump to 8 in. dib) = $\frac{0.02655\ D^2H^{1.13584}}{0.06869\ D^2H^{1.08516}}$ · Total Tree

Notes: D^2H = diameter outside bark (inches) at 4.5 ft above ground (dbh) squared times total tree height (feet).

Hardwood pulpwood trees are greater than 5.5 in. dbh.

Hardwood sawtimber trees are greater than 11.5 in. dbh.

2. Hard Hardwoods

Total tree = $0.14415\ (D^2H)^{1.032200}$

Pulpwood portion of main stem (stump to 4-in. dib) = $\frac{0.06393\ D^2H^{1.08601}}{0.13202\ D^2H^{1.04639}}$ · Total Tree

* All equations are for green weight of wood and bark in aboveground portions of tree. Foliage is included in overstory tree equations for pines. Foliage is not included in overstory tree equations for hardwards.

(Sheet 1 of 4)

Table V-3 (Continued)

Sawlog portion of main stem (stump to 8-in. dib)	$= \frac{0.044620\ D^2H^{1.110470}}{0.13202\ D^2H^{1.04639}}$. Total Tree

Notes: D^2H = diameter outside bark (inches) at 4.5 ft above ground (dbh) squared times total tree height (feet).

Hardwood pulpwood trees are greater than 5.5-in. dbh.

Hardwood sawtimber trees are greater than 11.5-in. dbh.

3. Pines

a. Loblolly Pine

Total tree	$= 0.11448\ (D^2H)^{1.03450}$
Pulpwood portion of main stem (stump to 2-in. dib)	$= \frac{0.14305\ D^2H^{0.99901}}{0.16228\ D^2H^{1.00404}}$ · Total Tree
Sawlog portion of main stem (stump to 6-in. dib)	$= \frac{0.07318\ D^2H^{1.06027}}{0.16228\ D^2H^{1.00404}}$ · Total Tree

b. Shortleaf and Pitch Pines

Total tree	$= 0.06175\ D^2H^{1.11931}$

(Continued)

Table V-3 (Continued)

Pulpwood portion of main stem (stump to 2-in. dib)	$= 0.06825\ D^2H^{1.08852}$
Sawlog portion of main stem (stump to 6-in. dib)	$= 0.05348\ D^2H^{1.09858}$

Notes: D^2H = diameter outside bark (inches) at 4.5 ft above ground (dbh) squared times total tree height (feet).

Pine pulpwood trees are greater than 3.5-in. dbh.

Pine sawtimber trees are greater than 9.5-in. dbh.

c. Virginia and White Pines

Total tree	$= 0.27765\ D^2H^{0.95706}$
Main stem (stump to 3-in. diameter outside bark (dob))	$= 0.11301\ D^2H^{1.03056}$

Pulpwood portion of main stem (stump of 4-in. dib) = Main Stem - 69.38 D - 3.53864 • Main Stem

(Continued)

(Sheet 3 of 4)

Table V-3 (Concluded)

Sawlog portion of main stem (stump to 6-in. dib)	= Main Stem - 69.38 D - 3.58539 · Main Stem

Notes: D^2H = diameter outside bark (inches) at 4.5 ft above ground (dbh) squared times total tree height.

D = diameter outside bark (inches) at 4.5 ft above ground (dbh).

Pine pulpwood trees are greater than 3.5-in. dbh.

Pine sawtimber trees are greater than 9.5-in. dbh.

Section B. Understory Trees

A. Hard Hardwoods

Total Tree = $5.06209\ (D^2)^{1.15258}$

B. Soft Hardwood

Total Tree = $4.41886\ (D^2)^{1.20114}$

C. Pine

Total Tree = $5.06209\ (D^2)^{1.15258}$

Notes: Same equation for hard hardwood and pines.

(Sheet 4 of 4)

Table V-4

Sources of Information for Biomass Equations used in the CALBIO Program

Clark, A., III, and J. G. Schroeder. 1977. Biomass of yellow poplar in natural stands in Western North Carolina. USDA S.E. Forest Expt. Sta., Res. Pap. SE-165.

Clark, A., III, and D. R. Phillips. 1979. Review Draft. Predicted abovestump total tree weight and volume of yellow-poplar, sweetgum, and white oak in the Georgia Piedmont. USDA S.E. Forest Expt. Sta., Athens, GA.

Clark, A., III, D. R. Phillips, and H. G. Hitchcock, III. 1979. Review Draft. Predicted total tree abovestump weight and volume of scarlet oak in the Tennessee Cumberland Plateau. USDA S.E. Forest Expt. Sta., Asheville, N.C.

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Phillips, D. R., and N. D. Cost. 1979. Estimating the volume of hardwood crowns, stems, and the total tree. USDA S.E. Forest Expt. Sta., Res. Note SE-276.

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Table V-5

Definitions of Biomass Categories

1. Pine Sawtimber: Wood and bark in main stem, from stump to 6-in. top diameter (inside bark), for pine trees greater than 9.5 inches dbh. Branches, foliage, and defects are not included.

2. Pine Pulpwood: Wood and bark in main stem, from stump or from sawtimber top to 2-in. top diameter (inside bark), for pine trees greater than 3.5 inches dbh. Branches, foliage, and defects are not included.

3. Pine Nonmerchantable: For pine trees greater than 3.4 in. dbh; it includes all aboveground tree components (wood, bark, branches, foliage, defects) not classified as pine sawtimber or pulpwood.

4. Pin Understory: Total tree for pines less than 3.5 in. dbh.

5. Hardwood Sawtimber: Wood and bark in main stem, from stump to 8-in. top diameter (inside bark), for hardwood tree greater than 11.5 in. dbh. Branches, foliage, defects, and noncommercial species are not included.

6. Hardwood Pulpwood: Wood and bark in main stem, from stump or from sawtimber top to 4-in. top diameter (inside bark), for hardwood trees greater than 5.5 in. dbh. Does not include branches, foliage, defects, or noncommercial species.

7. Hardwood Nonmerchantable: For hardwood trees greater than 3.4 in. dbh; it includes all aboveground tree components (wood, bark, branches, foliage, and defects) not classified as hardwood sawtimber or pulpwood. Noncommercial species are not included.

8. Hardwood Noncommercial: Includes all aboveground components (wood, bark, branches, foliage) of noncommercial hardwood species greater than 3.4 in. dbh.

9. Hardwood Understory: Total tree for hardwoods less than 3.5 in. dbh. Includes both commercial and noncommercial species.

10. Sawtimber: Combination of pine and hardwood sawtimber.

11. Pulpwood: Combination of pine and hardwood pulpwood.

12. Nonmerchantable and Noncommercial: Combination of pine and hardwood nonmerchantable biomass and hardwood noncommercial biomass.

13. Understory: Combination of pine and hardwood understory biomass.

14. Total: Combination of categories 1-9 (equivalent to combination of 10-13).

Table V-6

Green Biomass Weights in Russell Lake, by Vegetative Type and Density Class

Type	Density*	n*	Mean**	s**	$s_{\bar{n}}$**
Bottomland hardwood	Low	19	38.77	18.96	4.35
	Medium	24	75.49	18.43	3.77
	High	26	125.36	24.07	4.72
Upland hardwood	Low	26	37.58	17.72	3.48
	Medium	24	78.59	18.04	3.68
	High	7	121.09	29.44	11.13
Pine-hardwood	Low	11	41.31	16.50	4.97
	Medium	8	68.09	18.81	6.65
	High	4	104.56	32.90	16.45
Cutover	Low	25	17.95	13.65	2.73
Bluff hardwood	Medium	10	72.85	17.18	5.43
	High	10	118.88	20.05	6.34
	Total	20	95.86	18.62	5.89
Hardwood river edge		9	72.66	22.94	7.65
Total		203	68.75	18.97	1.33

* Green tons of biomass.
** Green tons of biomass per acre.

Table V-7

Weight in Green Tons of Flooded Biomass in Russell Lake, as Determined from Measurements Taken on 203 BAF 10 Points

Flooded Area	Vegetative Type/Density												
	Bottomland Hardwood			Upland Hardwood			Pine-Hardwood			Cutover	Bluff Hardwood	Hardwood River Edge	
	Low	Medium	High	Low	Medium	High	Low	Medium	High	(Low)	(Medium and high)	(Medium and high)	Totals
Beaverdam Creek	1,323.92	2,590.87	3,234.37	7,065.36	52,531.49	7,252.18	--	8,792.42	461.16	514.08	--	407.64	84,173.49
Lower Main Lake Basin	13,524.51	71,947.11	14,371.64	4,775.76	16,238.92	4,222.47	--	5,792.16	--	975.92	424.68	134.43	132,407.60
Rocky River	9,873.75	65,267.63	11,955.89	1,767.19	13,842.73	2,025.86	--	--	--	--	69.98	2,277.25	107,080.28
Savannah River	24,691.93	160,703.90	26,540.64	5,602.23	51,272.49	2,104.57	73.53	5,203.82	309.54	1,419.99	1,088.05	1,829.65	280,840.34
Total flooded forest area	49,414.11	300,509.51	56,102.54	19,210.54	133,885.63	15,605.08	73.53	19,788.40	770.70	2,909.99	1,582.71	4,648.97	604,501.71

Table V-8

Weight in Green Tons of Forest Biomass and Calculation of Submerged Biomass for the 13 Major Fish Tree Areas in Russell Lake

Fish Tree Area	Biomass green tons	Understory Biomass* green tons	Average Flooding Depth** ft	Percent of Overstory Biomass Below Surface†	Submerged Biomass green tons
Allen Creek	8,834.2	1,422.3	29	48.0	4,980.0
Bond Creek	5,745.5	913.5	24	39.8	2,836.6
Cedar Creek	1,788.4	284.4	19	31.5	758.2
Coldwater Creek	9,090.3	1,481.7	24	39.8	4,509.9
Dry Fork	4,114.4	720.0	19	31.5	1,789.2
Heardmont	16,813.3	2,824.6	34	56.3	10,700.2
Indian Creek	5,180.9	740.9	24	39.8	2,508.0
Latimer	8,048.8	1,247.6	24	39.8	3,954.5
Little Generostee	2,565.2	395.0	29	48.0	1,436.7
Nunley Branch	5,663.2	815.5	34	56.3	3,544.8
Pickens Creek	3,291.4	506.9	26	43.1	1,707.0
Rocky River	17,633.4	2,204.2	24	39.8	8,345.0
Total	96,105.7	14,627.8			51,148.8

* 12 tons/acre × number of acres in fish tree area = estimate of total submerged understory biomass.

** From contour lines, as shown on fish habitat area charts.

† Total - understory = overstory biomass. Below-surface overstory based on following proportion: 53% of total-tree biomass is contained in first 32 ft of stem height.

Table V-9

Weight in Green Tons of Submerged Forest Biomass in Russell Lake as of Initial Flooding, 1984

Flooded Area	Total Biomass green tons	Biomass in Fish Tree Areas green tons			Total Submerged Biomass* green tons
		Above-water	Submerged	Total	
(1)	(2)	(3)	(4)	(5)	(6)
Beaverdam Creek	84,173.5	8,231.5	14,245.0	22,476.5	75,942.0
Lower Main Lake Basin	132,407.6	0	0	0	132,407.6
Rocky River	107,080.3	13,382.7	12,299.5	25,682.2	93,697.6
Savannah River	280.840.3	23,342.7	24,604.3	47,947.0	257,497.6
Total	605,501.7	44,956.9	51,148.8	96,105.7	559,544.8

* Total submerged biomass (column 6) = total biomass (column 2) less above-water biomass in fish trees areas (column 3).

PART VI: IMPACT OF LEAF LITTER BREAKDOWN ON SOME WATER QUALITY CHARACTERISTICS OF RECENTLY IMPOUNDED RESERVOIRS*

Introduction

123. Impaired water quality conditions may occur immediately after the impoundment of a reservoir when terrestrial material is inundated rather than removed. The breakdown of forest vegetation, leaf litter, and soils may exert a significant demand on dissolved oxygen stores and provide a large internal source of nutrients and metals to the water column via leaching, microbial and invertebrate decomposition, and chemical interactions at the sediment/water interface. For instance, Kennedy and Nix (1986) report the occurrence of severe anoxia in a major portion of the hypolimnion of DeGray Lake, Ark., during the first several years of impoundment. Terrestrial vegetation was left within the basin of this system. Mouchet (1984) found that reservoirs in tropical regions which contained inundated vegetation experienced hypolimnetic anoxia and chemical stratification (i.e., elevated levels of ammonia, iron, manganese, and hydrogen sulfide). However, reservoirs cleared of vegetation and top soil prior to filling did not exhibit this trend. While the practice of site preparation (i.e., removal of vegetation and soils) has been recommended to improve water quality conditions of reservoirs (Benedetti and Roller 1962, Campbell et al. 1975), studies which quantify the impacts of inundated vegetation are needed (Gunnison et al. 1984, Mouchet, 1984).

124. Although tree biomass is more resistant to decomposition, leaf litter is highly labile and may have a significant impact on reservoir water quality. The impoundment of Southern Indian Reservoir, for instance, inundated 5.4×10^5 tons of black spruce needles (*Picea mariana*). Of this, 87.5 percent was broken down within 1 year, resulting in the potential release of large quantities of nutrients (Crawford

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* Part VI was written by William F. James, Environmental Laboratory, US Army Engineer Waterways Experiment Station, Vicksburg, Miss.

and Rosenberg 1984). These authors reported that the rate of organic carbon loss from pine litter breakdown was higher than the annual primary productivity measured during the first year of impoundment.

125. The breakdown of leaf litter proceeds as a series of processes which are regulated by the chemical composition of the leaf material and environmental conditions. Leaching is rapid during the early stages of breakdown, followed by fungal and bacterial invasion and macroinvertebrate consumption (Peterson and Cummins 1974). The rate of breakdown depends on a variety of factors, including temperature (Suberkropp, Godshalk, and Klug 1975; Barnes, Ovink, and Cummins 1978), dissolved oxygen, and nutrient availability for microbial metabolism (Kaushik and Hynes 1971, Triska and Buckley 1978, Triska and Sedell 1976, Howarth and Fisher 1976, Meyer 1980). These characteristics, therefore, result in a range of breakdown rates as a function of leaf type, susceptibility to microbial and invertebrate consumption, and environmental conditions. Differential rates of breakdown in a recently impounded reservoir could, therefore, have prolonged impact on dissolved oxygen and nutrient concentrations in the water column.

126. The impoundment of Russell Lake on the Savannah River resulted in the inundation of 3,490 ha of forested area containing tree bole, soils, and leaf litter. The objective of this study was to determine (a) rates of leaf litter breakdown using various species commonly found within the basin, and (b) the nutrient load and dissolved oxygen demand of this material during the first year of impoundment.

Methods

Leaf litter breakdown

127. Freshly abscised leaves collected from white oak (*Quercus alba*), hickory (*Carya* sp.), American beech (*Fagus grandifolia*), red oak (*Quercus rubra*), and short-needle pines were air-dried to a constant weight prior to incubation. In addition, a hardwood litter mixture (also refered to as detritus) was collected within the main basin of Russell Lake before filling. Ten grams of leaf material or litter

mixture were placed in 5-mm-mesh plastic bags (40 by 15 mm, dimensions) and four-replicate bags of each leaf type were secured to anchors. One anchor represented an individual sampling date; therefore, mechanical disruption of other deployed leaf bags during retrieval was prevented. All leaf bags were deployed on 15 February 1984, approximately 15 days after pool elevation reached 143.4 NGVD (i.e., 1.5 m below power pool elevation of 144.9 m NGVD).

128. Leaf bags were deployed on the lake bottom at the 3- and 28-m depths at the locations indicated in Figure VI-1. These depths represented the approximate upper and lower boundaries of the inundated forest. Conditions ranged from a high temperature and dissolved oxygen environment at the 3-m depth to a low temperature and anoxic environment at the 28-m depth. Replicate samples were retrieved from each station at intervals of 0, 14, 27, 98, 175, and 293 days, then placed in individual paper sacks and returned to the laboratory where they were gently washed to remove debris and sediment. Leaf material was oven-dried at 60° C for 5 days then reweighed to determine the percentage remaining.

129. Leaf material was ground in a Wiley Mill through a 40-mesh screen for chemical analysis. Subsamples were ashed in a muffle furnace at 550° C for 12 hr to determine ash-free dry weight (AFDW), a reflection of the amount of organic material. Total organic carbon was calculated as 47 percent of the AFDW (Howarth and Fisher 1976). Total phosphorus and total nitrogen were determined on a Technicon Autoanalyzer after a persulfate oxidation and reduction to nitrate with Devarda's alloy, respectively (Raveh and Avinimelech 1979).

130. Decay coefficients were calculated using a negative exponential model (Peterson and Cummins 1974) as,

$$\%R = e^{-kt}$$

where

%R = percent of initial material remaining at time t

k = decay coefficient

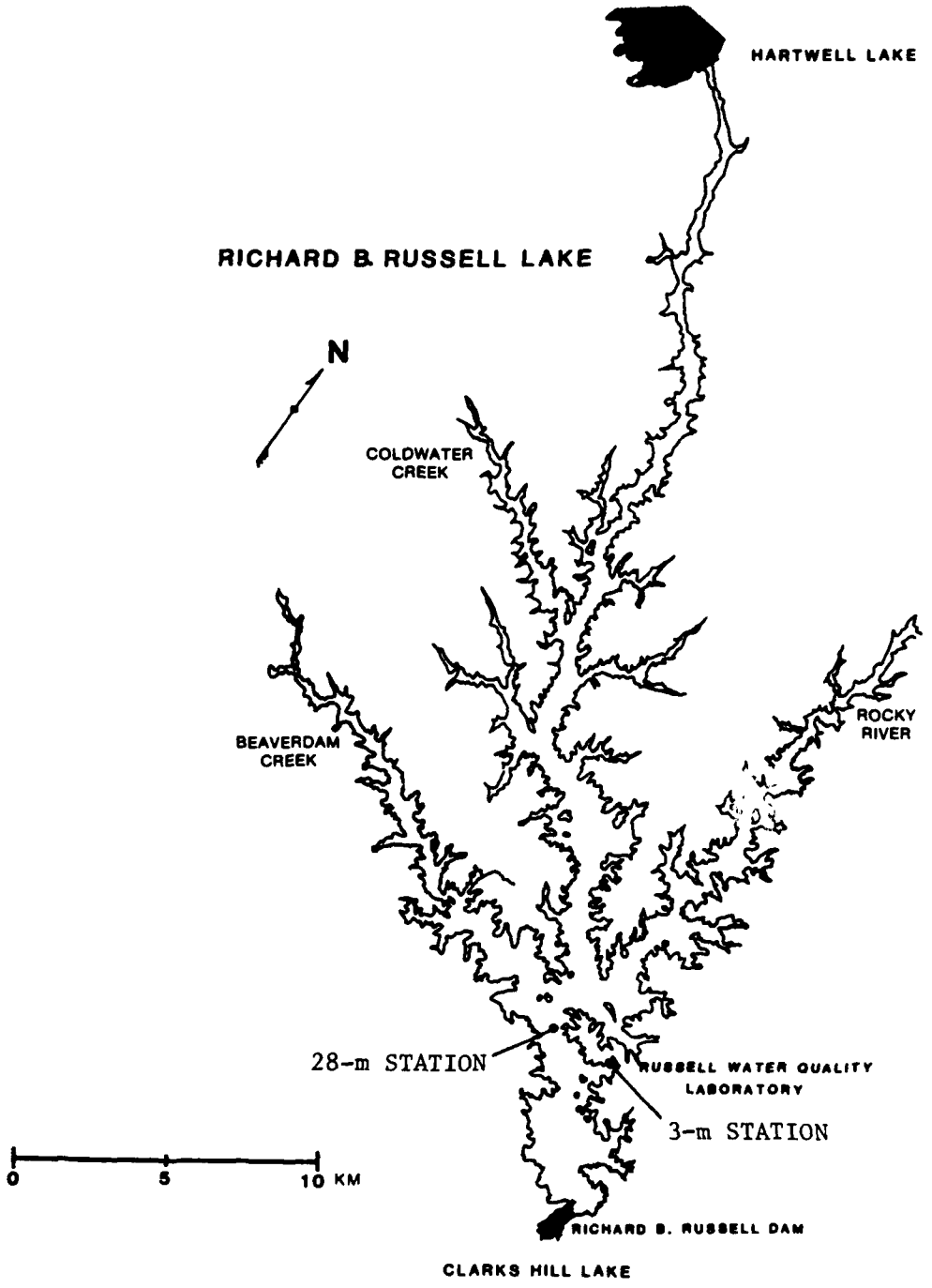

Figure VI-1. Location of study site at Russell Lake

Respiration rates

131. Rates of dissolved oxygen metabolism were determined for white oak and the hardwood litter mixture on each retrieval date at each station. Immediately after retrieval, 1-cm-diam leaf disks were bored from the center portion of randomly selected leaves. Ten disks were placed in each of four replicate darkened BOD bottles which were carefully filled with lakewater collected near the depth of the sample site. During the anoxic period, incubation water was collected in the upper, oxygenated zone of the hypolimnion to measure potential aerobic respiration rates on leaf litter. Dark bottles containing no leaf litter served as controls. Bottles were incubated on the lake bottom at each station for a maximum of 4 hr, then immediately fixed with Winkler reagents in the field for determination of dissolved oxygen (American Public Health Association 1980). Leaf disks were collected from each bottle for dry weight analysis. Preliminary experiments indicated that Winkler reagents had no effect on dry weight. Rates of respiration were calculated on a per gram AFDW and per square meter leaf disk basis.

Leaf litter biomass

132. To determine the impact of leaf litter decomposition on reservoir water quality characteristics, an estimate of leaf litter biomass left in the reservoir basin was required. This was determined indirectly using estimates of inundated tree biomass and total inundated forested areas (Part V). Since time constraints prevented the collection of litter samples, estimates of litter biomass on the forest floor were calculated as 1 percent of the average tree biomass per square meter.* It was assumed that (a) all leaves had fallen off the trees before inundation, (b) 1 percent of the tree biomass was a conservative estimate of the dry weight of leaf litter on the forest floor, and (c) tree biomass, and hence, leaf litter biomass, were homogeneous throughout all inundated forested areas.

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* Personal Communication, (1985), D. Phillips, Professor, Clemson University, Clemson, S.C.

133. The first assumption was validated since most of the forested area is hardwood deciduous and leaves had fallen off the trees before the initiation of filling in December. The second assumption appeared to be met since most of the inundated forest areas were of medium density and the mean tree diameter (dbh) was small. Studies done on saplings ranging from 3 to 11 in. dbh reported a dry weight leaf biomass equivalent to approximately 2 percent of the total tree biomass.* Thirdly, a preliminary survey of quadrate samples taken from various forest types indicated a marked similarity between calculated and actual measurements.

Russell water quality studies

134. In addition to estimates of leaf litter breakdown, extensive water quality monitoring was conducted during the first year of impoundment. These results, reported in James et al. (1985), were compared with results of the present study to evaluate the impact of leaf litter decomposition on reservoir water quality characteristics.

Results and Discussion

135. The loss of organic material (measured as AFDW) differed between stations and among leaf types (Figure VI-2). Weight loss due to leaching was not pronounced at either station during the first month of decomposition. However, short-needle pine and hickory lost considerable weight (38 and 18 percent, respectively) at the 28-m depth during this period. In general, breakdown for most species was more rapid at the 3-m depth. Hickory and white oak decomposed most rapidly at this depth with only 52 and 55 percent, respectively, remaining by December. Beech and red oak broke down more slowly while the hardwood litter mixture had an intermediate percentage (65 percent) remaining in December. Percentages remaining by December were much higher at the 28-m depth for all

* Personal Communication, (1985), D. Phillips, Professor, Clemson University, Clemson, S. C.

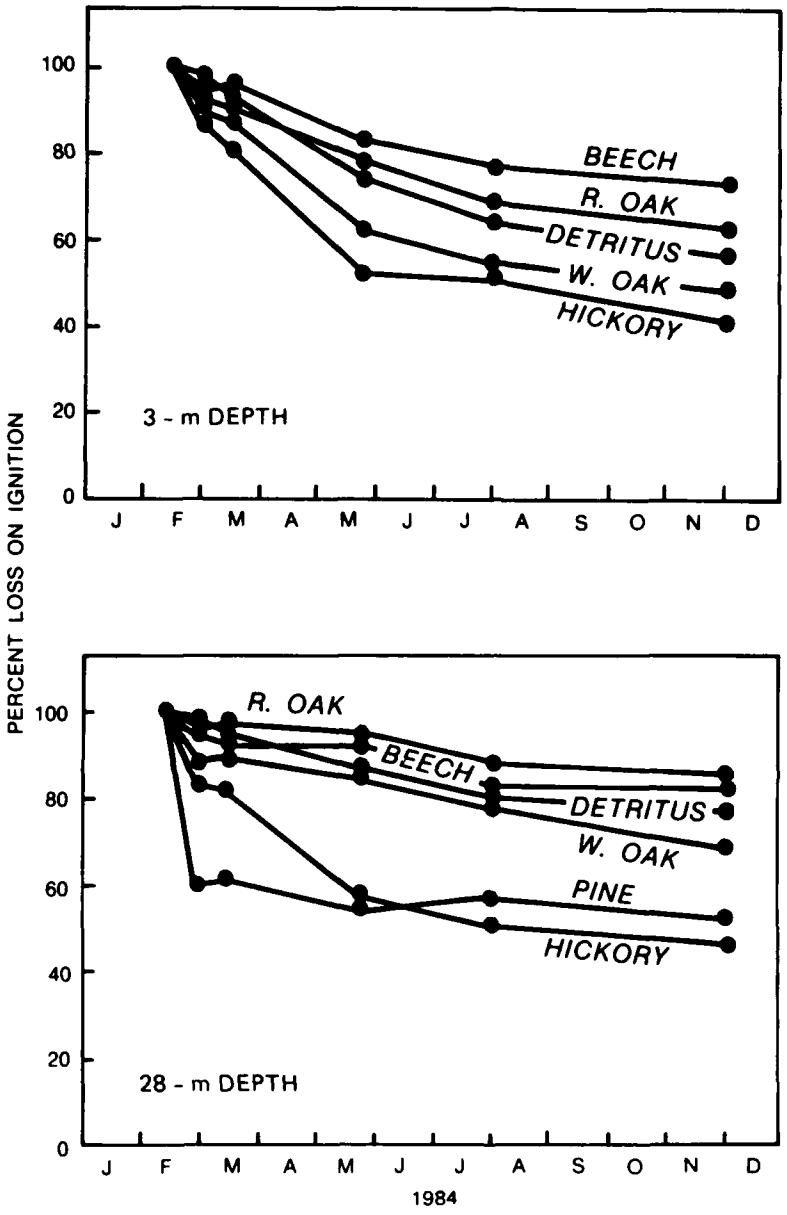

Figure VI-2. Changes in percent loss on ignition of leaf and detritus samples incubated at 3- and 28-m depths in Russell Lake

leaf types except hickory, indicating generally lower breakdown rates at this depth.

136. Variations in the leaf litter decay coefficient (k) further indicated the occurrence of a range of breakdown rates among leaf types and differences between depths (Table VI-1). Overall, decay coefficients were extremely low compared to those measured in streams (see Peterson and Cummins 1974). Hickory had the highest decay coefficient at both stations, followed by white oak and the hardwood litter mixture. Beech and red oak had the lowest decay coefficients. In addition, decay coefficients were significantly ($p < 0.005$) higher at the 3-m than at the 28-m depth for all leaf types except hickory. The estimated time for decay of 90 percent of the leaf material ranged from 2.18 years) for hickory to 5.73 years for beech at the 3-m depth. The deep samples had much higher decay times for all leaf types except hickory and ranged from 2.52 years for hickory to 15.8 years for red oak. The hardwood litter mixture had an intermediate decay time of 3.00 years at the 3-m depth and 7.01 years at the 28-m depth.

137. Between-depth differences in decay coefficients appeared to be related, in part, to differences in temperature and dissolved oxygen conditions (Figure VI-3). Temperature was low at the 28-m depth throughout the study period, ranging from 6.1° C in February to 14.2° C by November. Anoxic conditions were evident at this depth from May until November. High dissolved oxygen concntrations (i.e. > 5 mg/ℓ) were observed during most of the year at the 3-m depth, and temperature increased from 9.6° C in February to 24.0° C in early August. These differences probably had an influence on the metabolic activity of microbial communities colonizing the leaf litter and, therefore, the decay coefficients.

138. Rates of respiration, measured from white oak and the hardwood litter mixture, reflected these between-depth differences in the decay coefficient, as well as differences in the temperature and dissolved oxygen conditions. Areal and weight-based respiration rates exhibited similar seasonal trends (Table VI-2; areal-based respiration rates are not shown). In general, respiration rates for both leaf types

Table VI-1

Leaf Litter Decay Coefficients, Correlation Coefficients, and Estimated Time for 90-percent Loss of AFDW for Leaf Types Incubated at the 28-m and 3-m Depths

	28-m Depth			3-m Depth			
Leaf Type	k	r^2	90% Lost, yr	k	r^2	90% Lost, yr	Slope Test*
Hickory	0.0025	0.84	2.52	0.0029	0.86	2.18	N.S.
White oak	0.0010	0.86	6.31	0.0026	0.94	2.43	$p < 0.005$
Detritus	0.0009	0.88	7.01	0.0021	0.95	3.00	$p < 0.005$
Beech	0.0006	0.85	10.51	0.0011	0.90	5.73	$p < 0.005$
Red oak	0.0004	0.78	15.77	0.0014	0.65	4.51	$p < 0.005$
Short-needle pine	0.0015	0.40	4.21				

* Indicates significant differences between stations (N.S. = not significant).

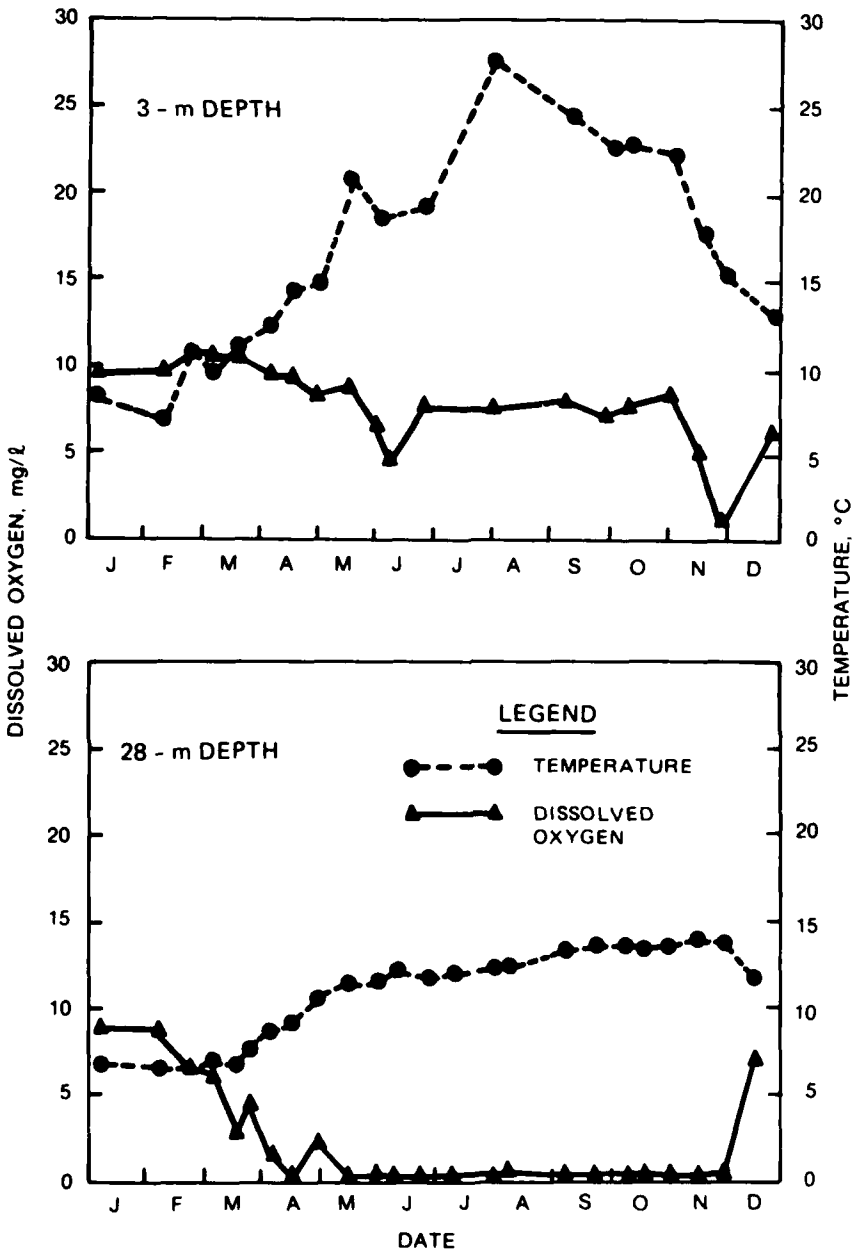

Figure VI-3. Changes in water column temperature and dissolved oxygen at each incubation depth

Table VI-2

Seasonal and Spatial Patterns in Respiration Rates Measured for White Oak and the Hardwood Mixture

Leaf Type	Date				
	1 March	15 March	14 May	9 August	8 December
			3-m Depth		
White oak	11.0 (1.1)	15.9 (3.1)	102.9 (16.4)	57.1 (7.1)	11.9 (2.0)
Hardwood mixture	19.7 (4.7)	14.5 (4.2)	63.2 (16.1)	50.1 (3.8)	7.6 (2.8)
			28-m Depth		
White oak	26.2 (7.2)	6.7 (1.8)	9.4 (2.8)	11.2 (1.2)	35.6 (6.2)
Hardwood mixture	15.5 (2.8)	4.2 (0.8)	18.3 (2.0)	10.8 (2.6)	29.0 (4.0)

NOTE: Values are expressed as milligrams O_2/gm AFDW per Day (± 1 standard error values).

were seasonally low at the 28-m depth and displayed minimal fluctuation. For example, rates ranged from 4.2 mg O_2/gm AFDW per day in March to 29 mg O_2/gm AFDW per day in November and were below 20 mg O_2/gm AFDW per day during the summer months. However, respiration rates increased markedly at the shallow station during the summer. Respiration rates ranged from 14.5 mg O_2/gm AFDW per day in March to over 50 mg O_2/gm AFDW per day in May and August for the hardwood litter mixture. Rates then declined in December during the period of cooler water temperatures. Differences in the rate of respiration were minimal between the hardwood litter mixture and white oak throughout the study.

139. The percent nitrogen content (i.e., gm N/gm AFDW) of the leaf material increased throughout the decay process for all leaf types at both depths (Figure VI-4). This trend was probably the result of retention of nitrogen on the leaves as microbial biomass. Percent nitrogen decreased initially due to leaching during the first month of decomposition for white oak and pine at the 28-m depth and for white oak and red oak at the 3-m depth. However, other leaf types displayed increases during this period. The maximum increase in percentage of nitrogen [(maximum percent N/minimum percent N) - 1] was highest for white oak (169), short-needle pine (164), and hickory (144 percent) at the 28-m depth.

140. Red oak and white oak displayed a maximum percent increase of 170 and 180 percent, respectively, at the shallow station. However, a good correlation was not evident between the maximum percent nitrogen increase and the decay coefficient. This might suggest a relationship between the rate of microbial conditioning (as measured by nitrogen retention on the leaves) and the decay rate.

141. A comparison of the percent nitrogen as a function of percent leaf material remaining was made to detect differences between leaf material in the same state of decay at the two depths (Meyer and Johnson 1983). In general, leaf litter in a similar state of decay retained more nitrogen at the 28-m depth than at the 3-m depth (Figure VI-5). Linear regressions of the percent AFDW remaining versus percent nitrogen revealed significantly ($p < 0.005$) higher coefficients at the 28-m depth

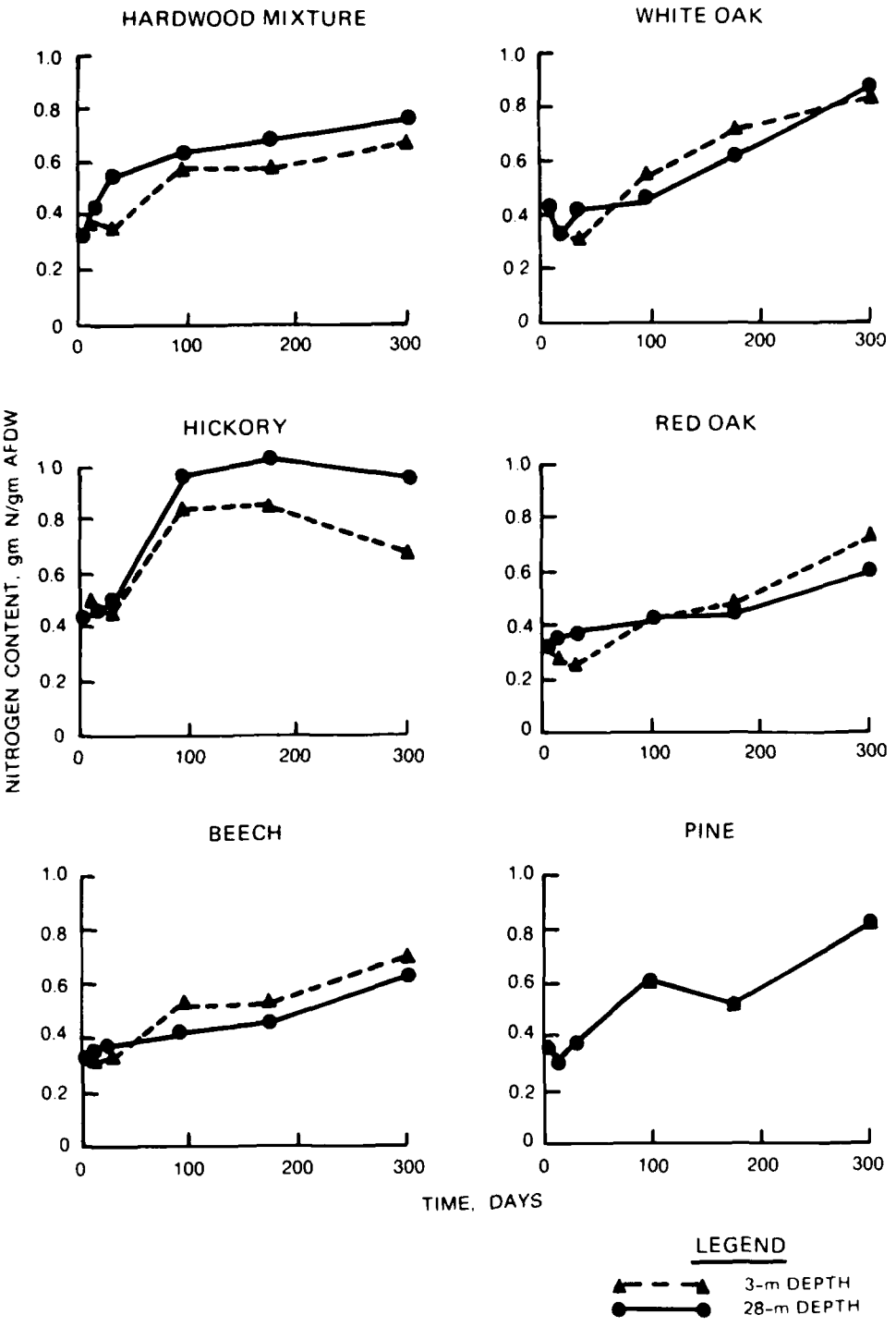

Figure VI-4. Changes in nitrogen content of incubated leaf samples

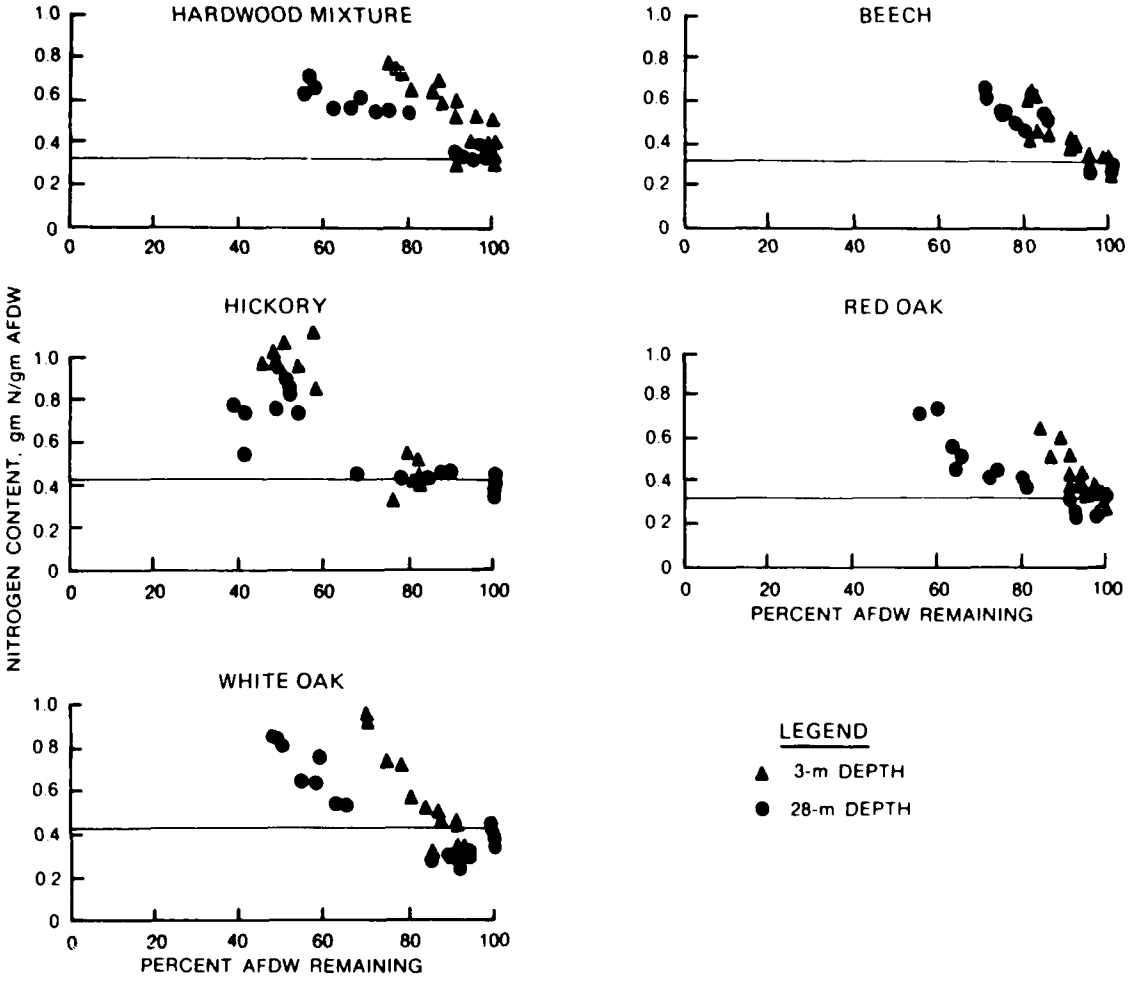

Figure VI-5. Nitrogen content of incubated samples as a function of percent AFDW. Horizontal line indicates average initial nitrogen content

for all leaf types except beech. This was unusual since leaf litter at the 3-m depth had higher decay coefficients, which would suggest greater microbial colonization and possibly higher nitrogen assimilation (hence, more nitrogen retention on leaves as microbial biomass).

142. Although more information is needed, differences in nitrogen retention may reflect differences in nitrogen metabolism between the two depths, since the 28-m depth was anaerobic while the 3-m depth was aerobic. The percent nitrogen coefficient ranged from 0.0246 percent N/percent AFDW for white oak to 0.0134 percent N/percent AFDW for the

Table VI-3

Percent Nitrogen and Percent Phosphorus Coefficients* for Various Leaf Types at the 3- and 28-m Depths

Leaf Type	Depth 28-M	Depth 3-M	Slope Test
	Nitrogen Coefficient		
White oak	0.0246	0.0117	$p < 0.005$
Red oak	0.0215	0.0109	$p < 0.005$
Hickory	0.0174	0.0074	$p < 0.005$
Beech	0.0149	0.0126	N.S.
Hardwood Mixture	0.0134	0.0073	$p < 0.005$
	Phosphorus Coefficient		
Leaf Type	28 M	3 M	Slope Test
Hickory	0.0034	0.0005	$p < 0.005$
White oak	0.0023	0.0003	$p < 0.005$
Hardwood Mixture	0.0014	0.0003	$p < 0.005$
Beech	0.0012	0.0006	$p < 0.005$
Red oak	0.0003	0.0005	N.S.

* Values were obtained from regressing percent AFDW remaining with percent nitrogen or phosphorus.

hardwood mixture at the 28-m depth (Table VI-3); however, there was no strong correlation between the decay coefficient and the percent nitrogen coefficient among leaf types.

143. Leaf material also exhibited a net increase in percent phosphorus (i.e., grams P/grams AFDW) during the latter stages of decay. However, phosphorus leaching was evident during the first month of decomposition for many leaf types (Figure VI-6). White oak litter exhibited the greatest decrease in percent phosphorus at the deep and shallow station (82 and 72 percent, respectively), followed by beech (45 and 45 percent, respectively), hardwood mixture (24 and 23 percent, respectively), red oak (31 and 5 percent, respectively), and short-needle pine (14 percent). Percentage increases after initial leaching were highest for hickory (302 percent) and hardwood mixture (124 percent) at the 28-m depth while red oak (96 percent) and hickory (83 percent) displayed highest increases at the shallow station. However, a clear trend was not evident among leaf types or between stations. As with nitrogen, leaf litter in the same state of decay had consistently higher percent phosphorus at the deep station (Figure VI-7). The exception was red oak. Although there was a range of values for the percent phosphorus coefficient (Table VI-3) among leaf types, significant relationships did not exist between the percent nitrogen and percent phosphorus coefficients or the decay coefficient.

144. The impoundment of Russell Lake resulted in the inundation of 3,490.1 ha of forested area containing 5.48×10^6 kg of green standing vegetation. Estimates of kilograms dry weight leaf litter and green weight tree biomass are shown for various locations in the reservoir in Table VI-4. Leaf litter remaining in the two major embayments, Beaverdam Creek and Rocky River, accounted for 14 and 17 percent of the total inundated leaf litter biomass, respectively. The upper and lower Savannah River Basin collectively accounted for 59 percent of the total biomass.

145. Although potential nutrient inputs from leaf litter were high, actual inputs to Russell Lake were minimal due to the low decomposition rates and the net retention of nitrogen and phosphorus on the

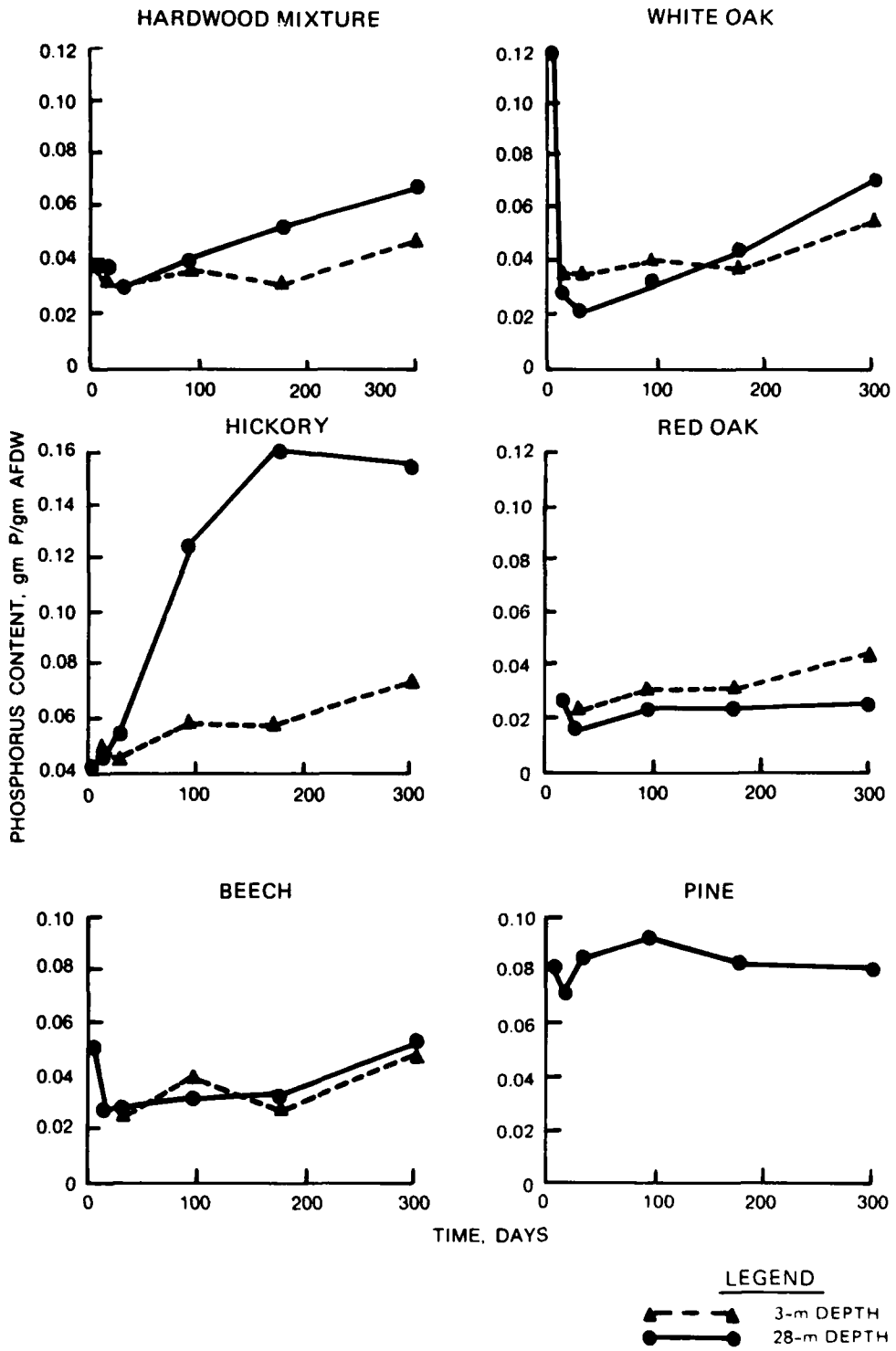

Figure VI-6. Changes in phosphorus content of incubated leaf samples

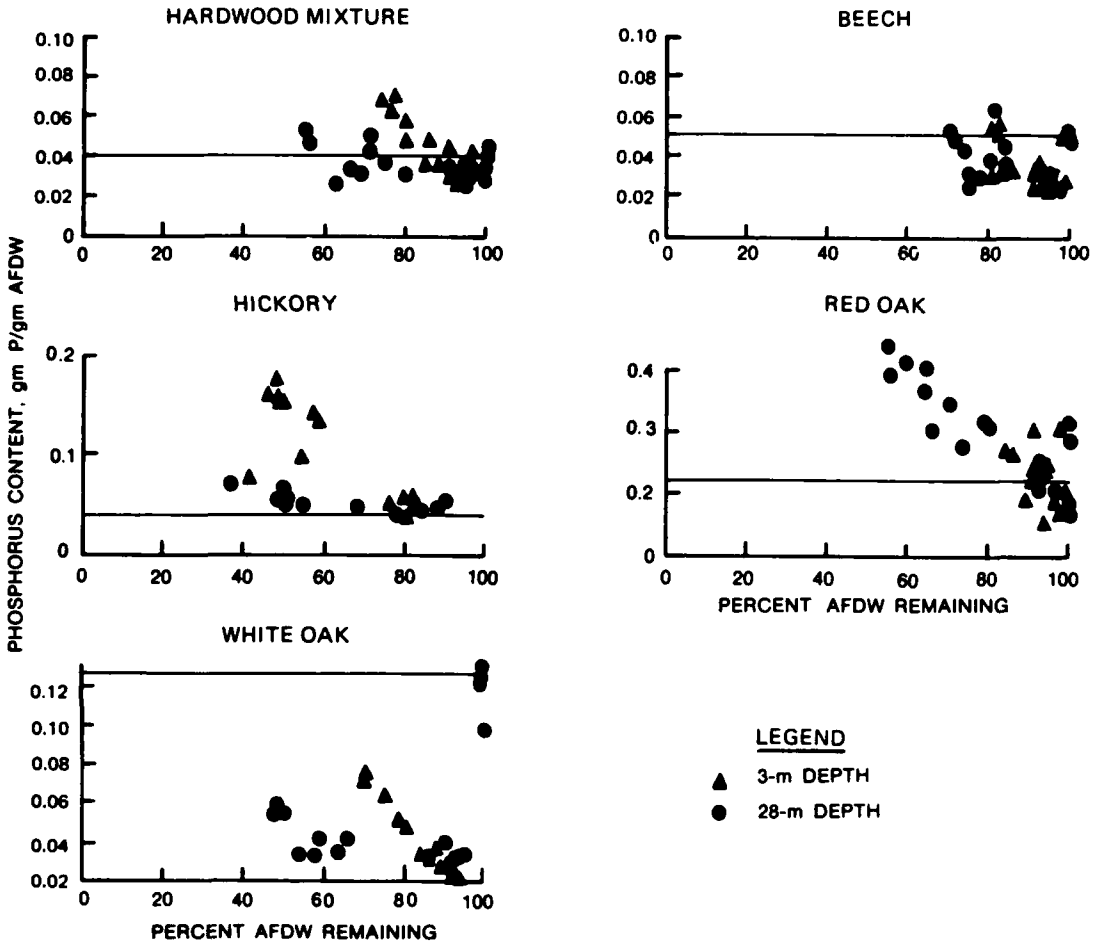

Figure VI-7. Phosphorus content of incubated samples as a function of percent AFDW. Horizontal line indicates average initial phosphorus content

leaf material. The hardwood litter mixture contained 47 percent C (percentage used by Howarth and Fisher 1976), 0.33 percent N, and 0.039 percent P. Complete breakdown of this material could potentially release 2.40×10^6 kg C, 1.68×10^4 kg N, and 2.00×10^3 kg P. Actual percentages of C, N, and P remaining through time were obtained from the

Table VI-4

Estimated Forested Areas, Green Weight of Tree Biomass,* and AFDW Leaf Litter for Various Areas of the Reservoir (see Figure VI-1)

Location	Forested Area, ha	Biomass Tree, kg	Leaf Litter AFDW, kg
Beaverdam Creek	477	7.6×10^7	7.0×10^5
Rocky River	601	1.2×10^8	8.9×10^5
Upper Savannah	1,630	9.7×10^7	2.3×10^6
Lower Savannah	781	2.5×10^8	1.1×10^6
Total	3,490	5.5×10^8	5.1×10^6

* Values taken from Part V.

hardwood litter bag data collected at the 28-m depth. These data indicated that only 5.45×10^5 kg C (i.e., 23 percent of the original weight) were lost during the first 270 days of inundation. Conversely, little release of N and P to the system was evident as the hardwood littermixture lost only 5.3 percent nitrogen and 10 percent phosphorus mass, as determined on a lakewide basis. These percentages were determined as the initial lakewide content of the leaves minus the content after 270 days (i.e., grams N or P/grams AFDW × total grams AFDW in lake).

146. These trends contrasted markedly with patterns observed in hypolimnetic total organic carbon, total nitrogen, and total phosphorus during the first year of impoundment, when pronounced concentration increases occurred during stratification (James et al. 1985). For example, total organic carbon concentrations at the bottom depth averaged 2.3 mg/ℓ on a lakewide basis in February 1984. By September, concentration ranged from 6.1 mg/ℓ at Station 60 to 3.2 mg/ℓ at Station 160, and the two major embayments (Stations 130 and 140) exhibited concentrations of 6.9 and 6.1 mg/ℓ, respectively. Total and dissolved nitrogen,

ammonia, and total and soluble reactive phosphorus also displayed concentration increases during summer stratification.

147. Net hypolimnetic accumulation of total organic carbon, total nitrogen, and total phosphorus in Russell Lake was determined for the stratified period and compared to variations in the nutrient content of the leaf litter calculated for the entire lake. During the decomposition process, losses of nitrogen and phosphorus from leaf litter could not account for the net increases observed in the hypolimnion (Table VI-5). The loss of organic carbon from the leaf litter was considerable and could account for 83 percent of the total organic carbon increases in the hypolimnion, assuming that the litter was converted entirely into fine particulate and dissolved organic carbon. However, the actual relationship between loss of organic carbon from the litter and accumulation in the hypolimnion could not be established because a major fraction of this particulate organic carbon could have been converted to CO_2 via metabolism and, hence, would not contribute to the organic carbon pool in the hypolimnion.

148. The dissolved oxygen demand created by decomposing leaf litter played an important role in hypolimnetic oxygen dynamics in Russell Lake. Estimates of leaf litter (i.e., grams per square meter; values from Table VI-4 divided by area) were multiplied by an average respiration rate of 15.56 mg O_2/gram AFDW per day (obtained from the hardwood litter mixture at the 28-m depth) to estimate the oxygen demand

Table VI-5

Change in Leaf Nutrient Content and Lakewide Hypolimnetic Accumulation

Nutrient	Change in Leaf Nutrient Content, kg	Hypolimnetic Accumulation, kg
Organic carbon	-5.5×10^5	6.6×10^5
Nitrogen	$+1.3 \times 10^4$	1.4×10^5
Phosphorus	$+6.2 \times 10^2$	1.2×10^4

exerted by leaf litter decomposition during the first year of impoundment. Areal estimates of leaf litter were similar for each region of the reservoir, averaging 145 gm/m^2, and areal respiration rates averaged 2,258 mg O_2/m^2 per day. Hypolimnetic dissolved oxygen demands in the reservoir were highest in the Beaverdam Creek (3,953 mg O_2/m^2 per day) and Rocky River (3,464 mg O_2/m^2 per day) embayments, and in the Upper Savannah River (3,537 mg O_2/m^2 per day). The demand exerted by leaf litter accounted for 58 percent of the total hypolimnetic demand in Beaverdam Creek, 67 percent in Rocky River and 63 percent in the Upper Savannah River. Thus, inundated leaf litter exerted a substantial and immediate demand on oxygen stores in the hypolimnion of Russell Lake.

149. Dissolved oxygen depletion and anoxia were severe during the first year of impoundment of Russell Lake as a result of the high dissolved oxygen demand. Anoxic conditions were evident from the dam to Station 160 and along a major length of the two embayment areas in September 1984 (James et al. 1985).

Conclusions

150. Water quality conditions were impaired in Russell Lake during the first year of impoundment due, in part, to a large pool of nutrients, metals, and oxidizable materials left within the basin prior to filling. Major sources of these materials included topsoils, tree boles, leaf litter, and external loads retained in the reservoir. The reservoir experienced extensive hypolimnetic anoxia and the buildup of elevated concentrations of organic carbon, nitrogen, phosphorus, iron, and manganese in the bottom waters in a large portion of the pool (James et al. 1985). This study was conducted to examine the influence of leaf litter decomposition on these water quality characteristics.

151. In general, decay coefficients reported for various leaf types in this study were low compared to those determined for similar leaf species in streams, rivers, and other lakes. For instance, decay coefficients for white oak ranged from only 0.0010 to 0.0026 day^{-1} at the deep and shallow stations, respectively, in Russell Lake. Studies

conducted with leaves of the same species incubated in Clayton Lake, Va. (Webster and Simmons 1978) and Augusta Creek, Mich. (Peterson and Cummins 1974) reported much higher decay coefficients (i.e., 0.0052 and 0.0056 day^{-1}, respectively). Differences between these studies may be related, in part, to the degree of mechanical disturbance and fragmentation, and to the presence of macroinvertebrates. Loss of leaf material by fragmentation would be lower in a reservoir than in a stream where flow velocities and mechanical disturbances are greater. In addition, macroinvertebrate colonization, which can enhance the rate of decomposition, was not detected for leaf bags incubated in Russell Lake. These factors may have contributed to the lower decay coefficients observed in Russell Lake.

152. Deciduous leaves differ in their rate of decay, resulting in a continuum of species-dependent decay coefficients (Peterson and Cummins 1974). These among-species differences are often related to the initial lignin and percent nitrogen content of the leaf material, although other unknown factors may also be involved.

153. A continuum of species-specific decay coefficients was also observed at both depths in Russell Lake for the leaf types measured. In general, hickory and pine decomposed most rapidly, followed by white oak, the hardwood mixture, red oak, and beech. These patterns further suggest a continuum in the influence of species-specific decay on reservoir water quality characteristics. More rapidly decomposing species would be expected to have a short-term impact, providing a more rapid pulse of nutrients and an initially high biological oxygen demand to the system. Leaves that decay more slowly might have a longer impact of less magnitude.

154. In addition to these characteristics a continuum of site-specific decay rates was also observed in Russell Lake. Leaf types decomposed more rapidly at the shallow depth than at the deep depth, and this pattern was strongly related to dissolved oxygen conditions at the two stations. The 3-m depth was subjected to an aerobic environment which would facilitate more rapid and efficient microbial processing than an anaerobic environment. Seasonal patterns in the respiration

rate on the leaf litter suggested that temperature also played an important role in the rate of decay. Leaves incubated at the 3-m depth exhibited elevated rates of respiration during the warmer summer period; rates then declined as water temperature cooled in December.

155. Although leaf litter represented a potentially large pool of nutrients which could be released to the water column, loss of nitrogen and phosphorus from leaves was minimal after the initial leaching period. For all leaf types, there was a general trend of increase in the percent nitrogen and phosphorus with time. Several studies have reported percent nitrogen content increases during the early stages of decomposition, presumably due to nitrogen retention on leaves in association with microbial biomass (Iverson 1975; Triska, Sedell, and Buckley 1975; Suberkropp, Godshalk, and Klug 1976). Meyer (1980) found a good correlation between percent phosphorus content and sediment accumulation on the leaves, suggesting that deposition may influence the percent phosphorus content. Although cases of nitrogen and phosphorus loss from leaves have been reported (Meyer and Johnson 1983), data collected from Russell Lake indicate that leaf litter decomposition probably played a minor role in hypolimnetic nitrogen and phosphorus dynamics during summer stratification. Rather, it appeared that other inundated materials acted as more important sources of nutrients to the hypolimnion (e.g., inundated soils; Gunnison et al. 1984).

156. The loss of organic carbon from the leaves could account for a substantial percentage of the total organic carbon increase in the hypolimnion during summer stratification. This, however, assumes that all the organic carbon lost from the leaves is dissolved or fine particulate organic carbon. However, microbial decomposition may account for most of the organic carbon weight loss after the initial leaching period, suggesting conversion of particulate organic carbon to CO_2 in both aerobic and anaerobic environments. Hence, leaf litter decomposition may have contributed little to the total organic carbon pool in the water column. More research is needed to clarify organic carbon dynamics on leaf litter during decomposition and fluxes of organic carbon from inundated leaves to the water column.

157. The dissolved oxygen demand created by decomposing 1 af litter was responsible for a substantial proportion of the total hypolimnetic dissolved oxygen demand in the two major embayments and in the Upper Savannah River. The percentage contributed by leaf litter decomposition ranged from 58 percent for Beaverdam Creek to 67 percent for Rocky River. By contrast, Gunnison et al. (1984), in a laboratory study conducted on representative soils from Russell Lake, reported an areal dissolved oxygen demand of only 184 mg O_2/m^2 per day. Thus, soils accounted for only 5 percent of the total hypolimnetic demand based on their study.

158. If it is assumed that the respiration rate on leaves remains constant (i.e., 2,258 mg O_2/m^2 per day) as the litter decomposes, the areal demand exerted by this material will naturally decline in subsequent years as the inundated biomass becomes lost via decomposition and permanent burial. For instance, using the decay coefficient 0.0009 day^{-1} obtained from the hardwood litter mixture, 72, 52, 37, and 27 percent, respectively, of the initial litter biomass will remain within the basin during the next 4 years. The dissolved oxygen demand exerted by leaf litter could, therefore, decrease from an estimated 1,624 mg O_2/m^2 per day in 1985 to only 604 mg O_2/m^2 per day by 1988 (demands calculated as 15.56 mg O_2/m^2 per day × percent remaining at time (t) × initial areal biomass of 145 gm AFDW/m^2). These projected demands have implications for long-term changes in the hypolimnetic dissolved oxygen demand and seasonal duration of anoxia in Russell Lake. In general, leaf litter decomposition will have less impact on the lakewide hypolimnetic demand in the following years.

159. Can site preparation improve water quality conditions during the early years of impoundment? These data suggest that removal of leaf litter could have reduced the dissolved oxygen demand in Russell Lake during the first year of impoundment. However, it appeared that leaf litter was not a significant source of nutrients to the water column. Anoxic conditions created by a high demand can, however, stimulate chemical interactions at the sediment/water interface, resulting in the release of reduced nutrients and metals from sediments to the water

column. Leaf litter decomposition could, therefore, have an indirect effect on the nutrient budget of a lake. More information is needed on the role of other components in the system such as soils, tree bole, and external loads to evaluate the usefulness and costs/benefits of site preparation with regard to water quality characteristics.

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PART VII: ESTIMATION OF NUTRIENT AND METAL RELEASE FROM RICHARD B. RUSSELL AND CLARKS HILL LAKE SEDIMENTS TO THE OVERLYING WATER*

Introduction

160. A study to predict water quality on the newly filled Richard B. Russell Lake was conducted under the Environmental and Water Quality Operational Studies Program (Gunnison, Chen, and Brannon 1983). Results of the study indicated that the decomposition of vegetation and soil organic matter would severely deplete dissolved oxygen and deteriorate the water quality in the reservoir. The predicted magnitude of the oxygen demand in lake water depended on the organic matter content and the residence time of water in the reservoir. The study also predicted that a severe water quality problem would exist in the newly filled Richard B. Russell Lake for several years until the labile vegetation within the inundated area was completely decomposed. However, the study was conducted under controlled conditions in the laboratory, and no evaluation of the effects of hydrodynamics on the dilution and transport of the materials was performed.

161. Field evaluation of the release rates of nutrients and metals from sediment to the Richard B. Russell and Clarks Hill Lake ecosystems is needed to verify these laboratory findings. Presented here are the results of an in situ study to ascertain the degree of correspondence between field observation and the release rates of reduced substances derived from laboratory studies.

Methods and Materials

In situ incubation

162. A short-term in situ study was conducted at Stations 30, 80, and 120 (see Figures I-2 and I-3) in the summer of 1984 to determine the

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* Part VII was written by Rex L. Chen, Environmental Laboratory, US Army Engineer Waterways Experiment Station (WES), Vicksburg, Miss.

release rates of nutrients and metals from the bottom sediment to the overlying water under anoxic conditions in Richard B. Russell and Clarks Hill Lakes. Sediments were collected with an Eckman dredge, and 250 ml was added to each 1-ℓ wide-mouth plastic bottle. The bottles were divided into two groups, filled with aerated bottom lake water, capped without trapping air bubbles, and returned to the bottom of the lake at the respective stations. Water samples were collected periodically from the incubated bottles. One group of bottles was sampled periodically for 2 days after the experiment was initiated to obtain data on short-term incubation. The other group of bottles remained on the bottom of the lake for 21 days before collection of water samples was begun. The water samples were filtered through a 0.45-μ millipore filter, and the release rates of specific chemicals from the sediments were estimated by performing a linear regression analysis of mass release per unit area (milligrams per square meter) versus time (days).

Laboratory model system

163. Sediments collected from Stations 30, 80, and 120 were shipped to the WES Environmental Laboratory for flux estimation. The laboratory study involved the use of 14- by 88-cm acrylic columns equipped with glass diffusers placed 5 cm above the sediment/water interface. The dissolved oxygen concentration in the overlying water column was depleted by purging with nitrogen gas. To each column was added 4 ℓ of sediment and 8 ℓ of deionized water; a wax cap was placed on the top of the water column and sealed with mineral oil to prevent the entrance of oxygen from the atmosphere. The sediment/water columns were placed in a temperature-controlled chamber and held at 20° C in the dark for 45 days so that the anoxic characteristics of a packed sediment column would develop. An interstitial water profile sampler, constructed using dialysis membrane filled with distilled water, was vertically inserted into the sediment and was not disturbed for 10 days. The interstitial water samples were filtered, and concentrations of dissolved forms of ammonia, orthophosphate, iron, and manganese were determined in accordance with the methods recommended by the US Environmental Protection Agency (Ballinger 1979). The concentration gradient data were

incorporated into a modified of Fick's diffusion equation (Berner 1971) for estimating the flux rates of specific chemicals.

Results and Discussion

Field evaluation

164. An in situ study was conducted on Richard B. Russell and Clarks Hill Lakes to evaluate the fate of nutrients and metals in the water column. Results of the study showed that the sediments of these reservoirs would have great impact as a source of nutrients and metals for the hypolimnetic water during the period of thermal stratification. Release of specific chemicals from sediment was significant and largely dependent on the sediment characteristics. Concentration changes of soluble nutrients and metals in the overlying water during a short-term in situ incubation are summarized in Table VII-1. Stratification developed at all three sites, and accumulation of the chemicals was detected in anoxic waters during the study. Dissolved oxygen concentrations in the bottom waters at Stations 30, 80, and 120 were observed to be 1.9, 0.1, and 0.1 µg/ml, respectively. The trace amounts of dissolved oxygen that were detected at Station 30 (Clarks Hill Lake) during the stratified period were due mainly to a strong interflowing current.

165. A short-term in situ incubation does not provide enough time to allow sediment to deplete totally the dissolved oxygen in overlying water within an enclosed sediment-water system. Only a minor change in concentrations of soluble nutrients and metals was observed after 2 days of incubation under in situ conditions. A small accumulation of the reduced substances in the oxic overlying water is probably due to immobilization of these substances by oxidation processes.

166. For calculating release rate of chemicals from the sediment to the overlying water, the sediment-water bottles were returned to the bottom of the reservoir and preincubated under anoxic hypolimnetic water for 21 days to allow the system to become anoxic. After the water inside the bottles became anoxic, periodic sampling was conducted to evaluate the release rates of the constituents of interest. With the

Table VII-1

Changes in Nutrient and Metal Concentrations in the Overlying Water During in Situ Study at Richard B. Russell and Clarks Hill Lakes, Summer 1984

Incubation Time, days	Concentration, µg/ml			
	NH_4-N	Ortho-P	Fe	Mn
		Station 30		
0	1.073	0.004	2.517	3.531
1	1.554	0.009	2.680	4.429
2	1.614	0.019	3.815	4.773
		Station 8C		
0	2.825	0.045	12.665	1.258
1	3.232	0.092	13.303	1.092
2	3.877	0.088	15.390	1.350
		Station 120		
0	0.103	0.005	0.157	0.140
1	0.174	0.007	0.445	0.213
2	0.186	0.035	0.549	0.244

exception of sediments from Station 120 (newly inundated red clay with a low oxygen demand), concentrations of the specific constituents increased with increased incubation time (Table VII-1). Only minor release of reduced substances from the sediment to the overlying water was detected at Station 120.

Flux rates of nutrients and metals

167. The membrane may be considered to provide relatively insignificant resistance to specific diffusion after the system was equilibrated for 10 days (Hesslein 1976). The data indicated that relatively strong concentration gradients exist with respect to sediment depth, with the exception of coarse-textured, waterlogged soil (red clay) from Station 120. Chemical concentration in the interstitial water increased with increasing sediment depth. The vertical

distribution of the concentration gradient supports the upward movement of specific chemicals that were transported across the sediment/water interface into the overlying water.

168. The fluxes of dissolved ammonium, orthophosphate, iron, and manganese in the interstitial water were calculated based on the vertical distribution profile of interstitial constituents, using the following modification of Fick's equation:

$$J = -\phi * D * (dC/dZ)$$

where:

J = flux, $mg/cm^2 \cdot sec$

ϕ = the sediment porosity

D = the whole sediment diffusion coefficient, cm^2/sec

dC/dZ = the concentration gradient in interstitial water, mg/cm^3 per cm

169. Porosity and interstitial water concentrations of ammonium, orthophosphate, iron, and manganese measured in the sediments of the Richard B. Russell and Clarks Hill Lakes are presented in Table VII-2. Results of the sediment core interstitial water analyses for selected nutrients and metals are illustrated in Figure VII-1. These data indicate that relatively strong concentration gradients existed with respect to sediment depth at Stations 30 and 80.

170. Bulk sediment diffusion coefficients were estimated using the diffusion coefficients in water at infinite dilution at 18° C of Li and Gregory (1974) and the relationship $D = D_1 \times \phi^2$ to obtain an estimate of the bulk sediment diffusion coefficient (Lerman 1979, Watanabe and Tsunogai 1984). Values of D_1 used in calculating predicted fluxes for ammonium, orthophosphate, iron, and manganese were 1.45, 0.62, 0.50, and 0.50 cm^2/day, respectively. Fluxes of ammonium, orthophosphate, iron, and manganese estimated using Fick's equation and appropriate sediment and interstitial water data are presented in Table VII-3.

Table VII-2

Porosity and Concentrations of Interstitial Water Constituents in Richard B. Russell and Clarks Hill Lake Sediments

Station	Bulk Density g/ml	Porosity %	NH_4-N µg/ml	Ortho-P µg/ml	Fe µg/ml	Mn µg/ml
30	1.118	33.1	4.84	0.12	33.07	19.60
80	1.363	60.1	7.87	0.06	60.15	18.17
120	1.694	38.8	0.03	0	0	0

171. Fluxes of all parameters measured in this study using 25-cm sediment columns were higher than the values calculated in a previous chamber study in which the maximum depth of sediment used was 10 cm (Chen, Brannon, and Gunnison 1984). In addition to diffusion, various complex environmental factors, including microbiological activities and physical and biological turbation, can also affect the flux rates of chemicals from the sediment to the overlying water without a significant change in the pore water concentration gradient.

Conclusions and Summary

172. Comparison of these calculated (laboratory) flux values with observed (field) flux values indicated that, with the exception of Station 120, the flux values obtained from these studies were in good agreement.

173. Due to the complexity of environmental factors and the physical and chemical characteristics of sediment affecting the release rates of reduced constituents across the sediment/water interface (Berner 1971, Klump and Martens 1981), there exists no single procedure for selecting anaerobic release rates. The accumulation of phosphates in the sediment is highly dependent on the period of stratification in the bottom water and decomposition rate of organic matter at the sediment/water interface. Nevertheless, application of the interstitial

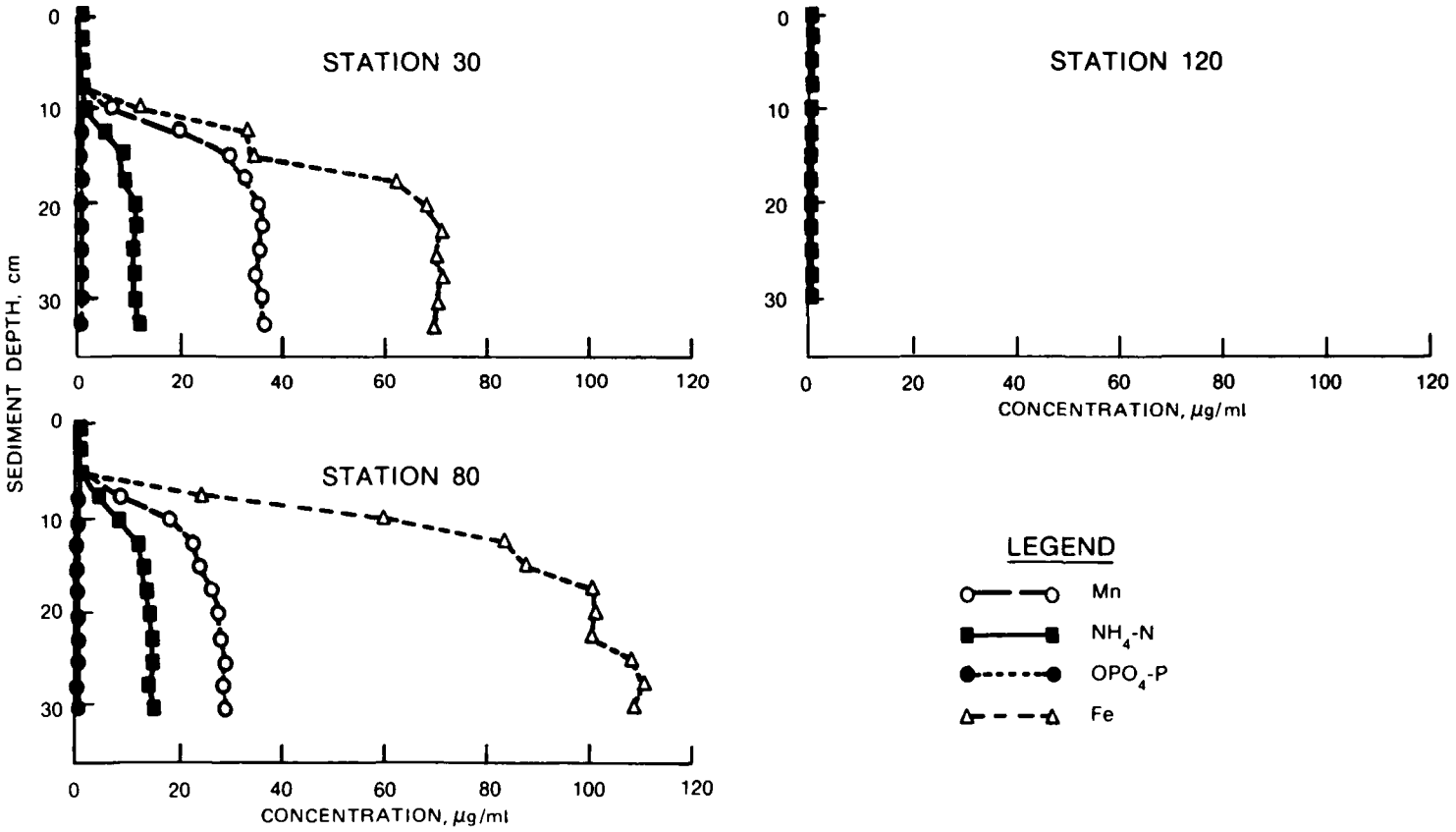

Figure VII-1. Vertical distribution of interstitial constituents

Table VII-3

Fluxes ($mg/m^2/day$) Calculated from Modified Fick's Equation

Station	NH_4-N	Ortho-P	Fe	Mn
		Laboratory Study		
30	11.3	0.8	71.6	36.0
80	14.8	0.2	110.6	29.1
120	0	0	0	0
		Field Study		
30	39.1	0.7	40.6	75.9
80	16.1	3.4	100.2	2.3
120	5.6	1.2	24.2	6.2

water device for directly (in situ) measuring the vertical concentration gradients from sediment cores may provide a meaningful approach for estimating the release rates of reduced substances in reservoirs.

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PART VIII: EPR ANALYSIS OF THE OXIDATION OF MANGANESE IN RICHARD B. RUSSELL LAKE*

Introduction

174. The accumulation of reduced metals such as manganese (Mn^{+2}) in anaerobic hypolimnia of lakes and reservoirs is a common phenomenon. The development of these anaerobic conditions in Richard B. Russell Lake (RBR) has already been observed (James et al. 1985). The operation of the oxygenation system will assist in the oxidation of the manganese, but the ultimate effectiveness of this system is not known at this time.

175. In natural environments, elements such as manganese are commonly found in the complexed, colloidal or particulate organic constituents. Such associations may be even more common in systems such as RBR where large amounts of organic material have been flooded. While the oxidation of manganese is normally quite slow, its complexation with organic material will undoubtedly retard the oxidation even more. Knowledge of the concentration of reduced manganese in RBR, both dissolved and in particulate matter, and the rate at which it can be oxidized are necessary in order to determine the extent of the problem and the effectiveness of its solution.

176. The objective of this study was to develop a method for determining the manganese dynamics in the reservoir, as well as for evaluating the efficiency of the oxygenation system for minimizing the amount of reduced manganese in RBR and its tailwater. We have shown that electron paramagnetic resonance (EPR) spectroscopy can be used to measure reduced manganese in RBR and to determine the rate of oxidation of the manganese when it is in an oxygen-enriched environment.

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* Part VIII was written by Raymond C. Turner, Department of Physics and Astronomy, Clemson University, Clemson, S. C.

Methods and Materials

177. EPR spectroscopy is particularly effective in determining the presence of reduced manganese both in solution and in solid form. This technique has been used to determine the concentration of reduced manganese in both dissolved and particulate fractions of samples taken from RBR. All EPR measurements were performed on a Varian E3 EPR Spectrometer and were made with the samples at room temperature. Initial measurements were made with the water samples in capillary tubes, but this was found to be too inaccurate. All subsequent measurements were made with the samples in a special cell designed for use with aqueous solutions.

178. The concentration of the reduced manganese in the water samples was determined by comparing the intensity of the EPR absorption signal of the samples with that of a standard solution. The standard solutions were prepared by dissolving known amounts of $MnCl_2$ in distilled water, and then stabilizing the solutions with 0.1-percent nitric acid.

179. The water samples from RBR were taken above and below the Russell Dam using standard sampling procedures. Precautions were taken to minimize the interaction of the samples with air. All samples were stored on ice in the dark prior to the EPR measurements. Unfiltered samples were taken as drawn from the lake, and contained both dissolved manganese and manganese in the particulate matter. Filtered samples were passed through a 0.45-μ filter in order to remove most of the particulate matter. These samples contained primarily dissolved manganese. A few of the samples were acid-stabilized in order to minimize the oxidation of the manganese.

180. The dynamics of oxidation of the manganese in RBR samples was determined by saturating the solution with dissolved oxygen and measuring the concentration of reduced manganese as a function of time. About 250 ml of the sample was placed in a gas washing bottle, and a sufficient amount of oxygen was bubbled through the water to maintain oxygen saturation. Small samples were then drawn periodically for

measurement by EPR. Samples that were not to be measured immediately were stored in a refrigerator in the dark.

Results and Discussion

181. A sample EPR spectrum obtained from RBR water is presented as Figure VIII-1. This six-line spectrum is what is expected from reduced manganese. This spectrum may be compared to that obtained from our standard 1 mg/ℓ reduced manganese sample, shown in Figure VIII-2. Only the gain was changed in measuring the two spectra. This shows that reduced manganese in RBR lake water can be observed by EPR; further, its concentration can be determined by comparing the EPR signal intensity to that of a standard sample. It was also found that the same EPR spectrum was obtained whether the sample was filtered or unfiltered, or had acid added as a stabilizer. The reduced manganese can be measured both in the dissolved state and in the particulate matter.

182. Relative concentrations of reduced manganese are more accurately determined by careful measurement of one of the six spectral lines, rather than by measuring the entire spectrum. It was found ultimately that the highest field line was subject to the least interference from other signals, and this line is now used for concentration determinations. The spectrum for this high field line for the 1 mg/ℓ standard sample is shown in Figure VIII-3. Using this spectral line gives a detection sensitivity of 0.05 mg/ℓ for routine measurements. This sensitivity can be improved, if necessary, by noise-averaging techniques.

183. In order to determine the optimum sample handling technique, four different water samples were prepared and their EPR spectra were measured as a function of time. Two samples each of filtered and unfiltered water were drawn; one of each was stabilized with acid. All samples were refrigerated in the dark until their EPR spectra were measured. The results of these measurements are shown in Figure VIII-4. They show essentially no change over a month's time in the reduced manganese concentration for the filtered sample and for the

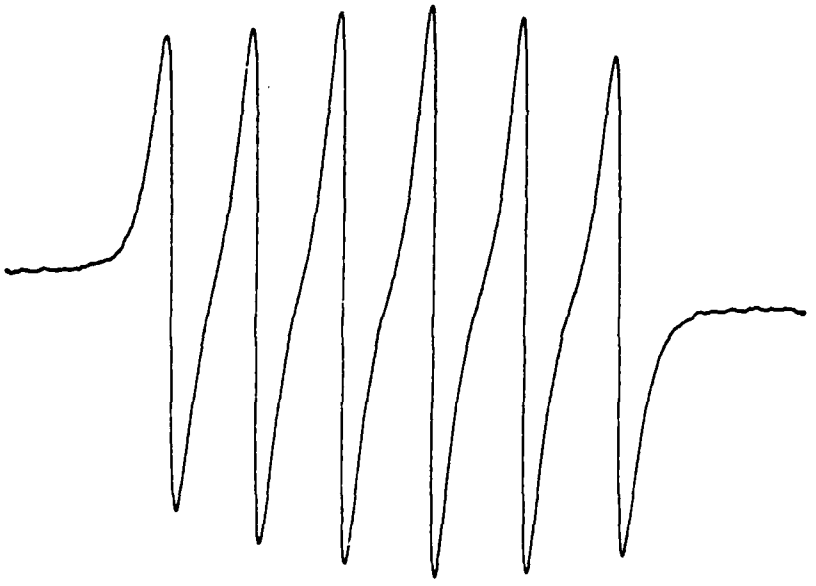

Figure VIII-1. An EPR spectrum of RBR lake water. Sample was obtained from a depth of 39 m at Station 100 on 27 September 1984

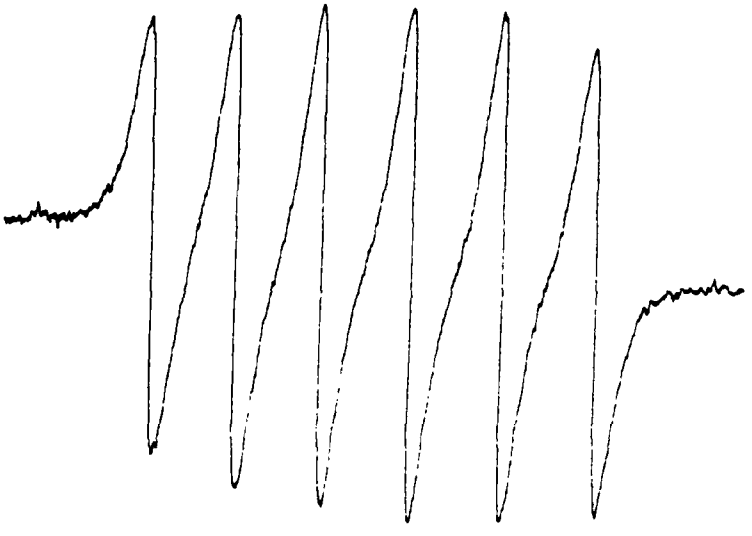

Figure VIII-2. An EPR spectrum of reduced manganese in water having a manganese concentration of 1 mg/ℓ

Figure VIII-3. An EPR spectrum of the high field manganese line of a standard solution containing 1 mg/ℓ manganese

acid-stabilized samples. Only the unfiltered, unstabilized sample oxidized significantly, and then only after about a week. The scatter in the data is probably due to the measuring techniques used at that time. The solid curve, which corresponds to an exponential decay with a time constant of 25 days, is shown for reference.

184. In addition to the above measurements, a portion of each of these samples was also maintained at ambient temperature and light (but still not exposed to air). The results of measurements on these samples are shown in Figure VIII-5. Once again, the acid-stabilized samples showed essentially no change, while both the unstabilized samples oxidized. The solid curve in this case corresponds to an exponential decay with a time constant of 15 days. The experimental significance of this is that no particular precautions are necessary when measuring the reduced manganese concentrations in acid-stabilized samples, but unstabilized samples should be measured within several days, even when stored refrigerated in the dark.

185. The significance of the results concerning manganese dynamics is that in the absence of oxygen, even at room temperature in the light,

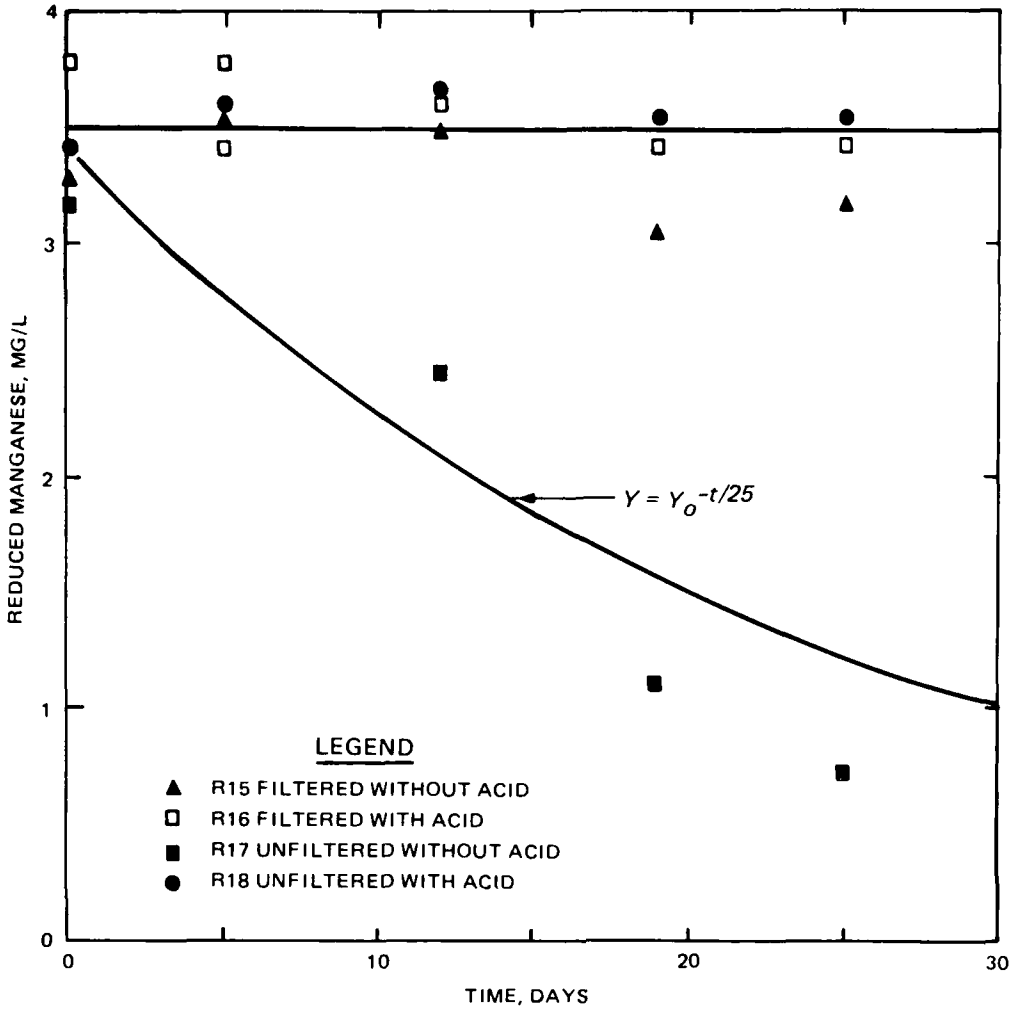

Figure VIII-4. Changes in reduced manganese concentration as a function of sample handling technique and holding time. All samples were refrigerated in the dark

oxidation apparently occurs with a time constant of many days. It should be noted that while these are preliminary results, they are probably at least qualitatively correct.

186. The results of measurements of oxidation of the manganese in lake water but in an oxygen-enriched atmosphere are shown in Figures VIII-6 and VIII-7. The sample used for Figure VIII-6 was filtered, while the sample used for Figure VIII-7 was unfiltered. It should be noted that when these experiments were performed, experimental difficulties were encountered and measurements could not be made at that time. Rather, the samples were stored in the refrigerator after they

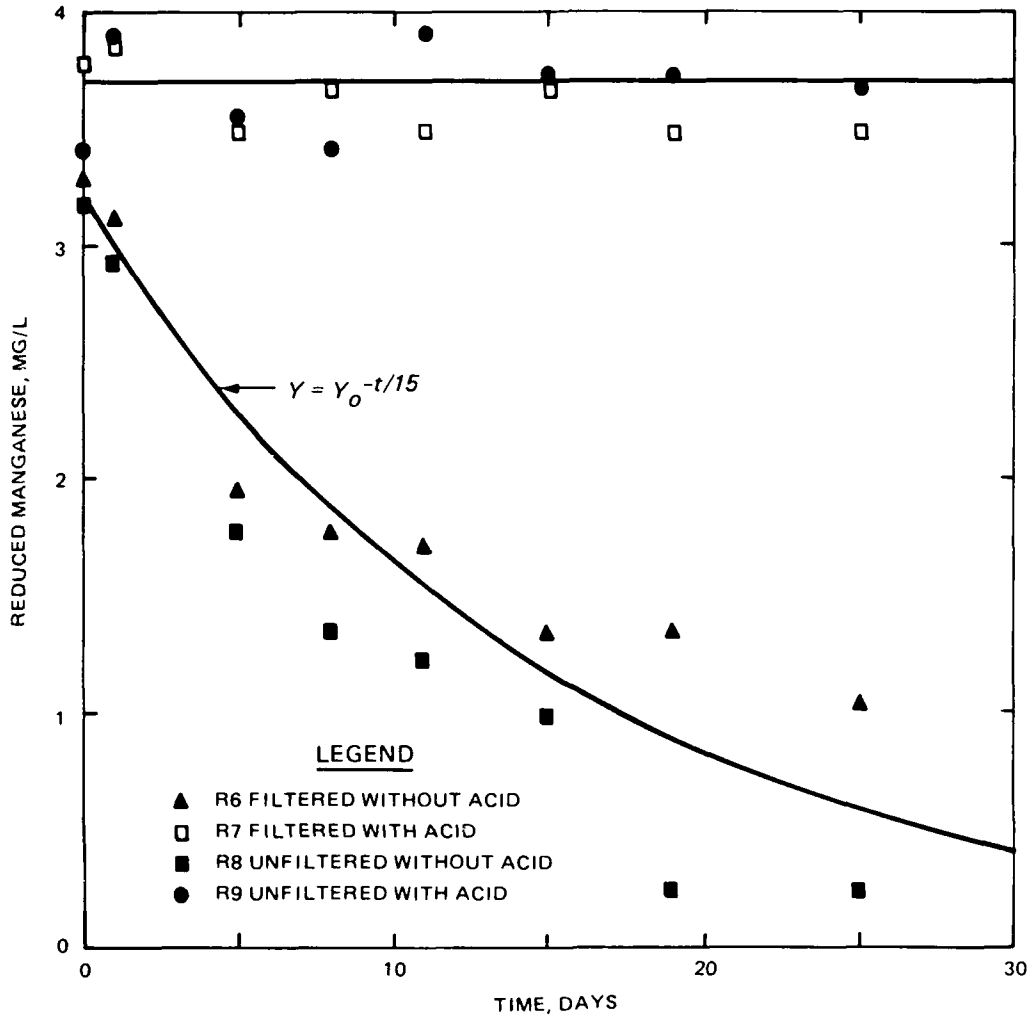

Figure VIII-5. Changes in reduced manganese concentration as a function of sample handling technique and holding time. All samples were held at room temperature in the light

were drawn, and they were measured a couple of months later. Nonetheless, the effect of exposure time to oxygen saturation is quite apparent. The solid curves in the figures correspond to exponential decay with time constant of 1.5 hr. These results would imply that in oxygen-saturated water (at about 23° C and in light), the oxidation of the manganese takes place with a time constant of about 2 hr. Two additional items about these results should be noted. One is that measurements at a different time on the same samples but stored at room

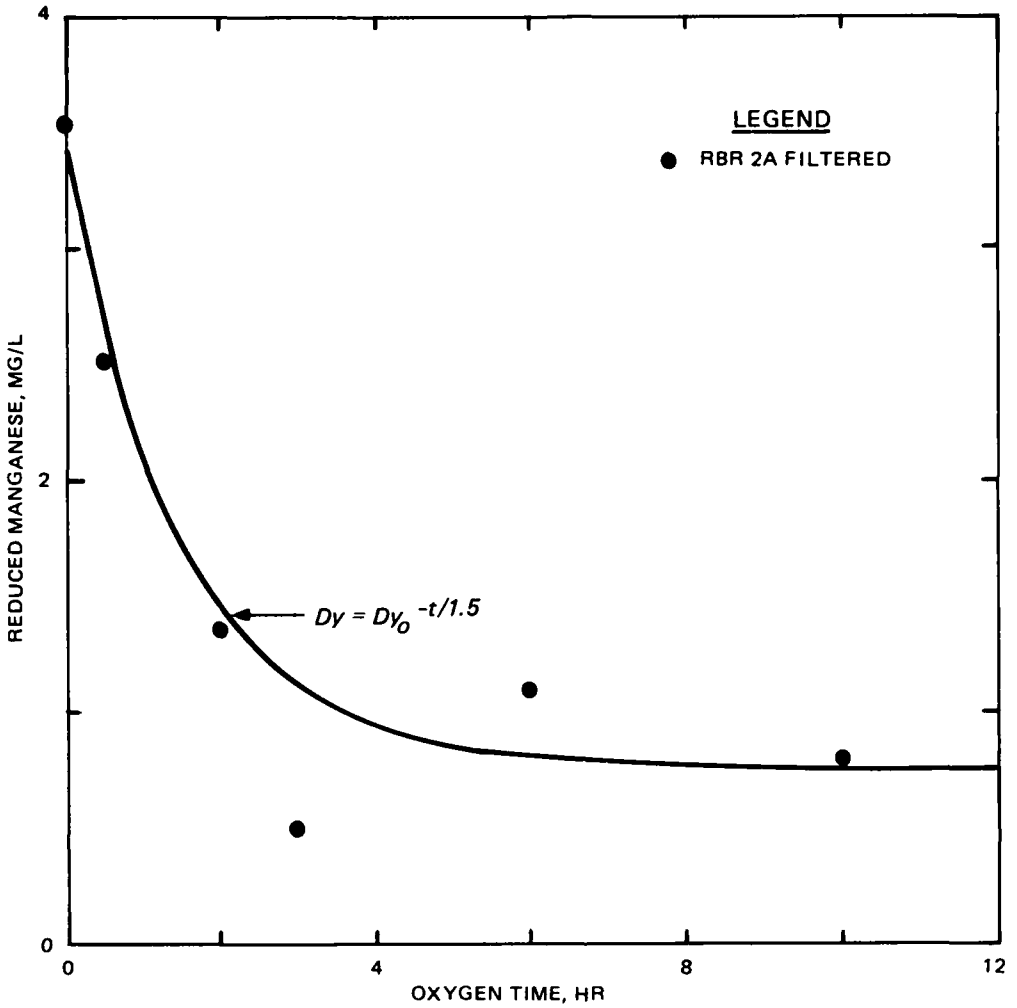

Figure VIII-6. Change in the reduced manganese concentration of a filtered sample as a function of duration of oxygenation of the sample. Solid curve corresponds to an exponential decay with a rate constant of 1.5 hr. Dy_o and Dy are the initial concentration and the concentration at time t

temperature indicated a time constant of about 3 hr. Thus, the actual value of the time constant should not be taken too literally at this point, but is probably approximately correct. The second item to note is that the exponential decay in the curves is not to zero, but rather to a concentration of about 0.5 mg/ℓ reduced manganese. This may be an indication that a small portion of the manganese oxidizes much more slowly. Although this could be due to the manganese in particulate

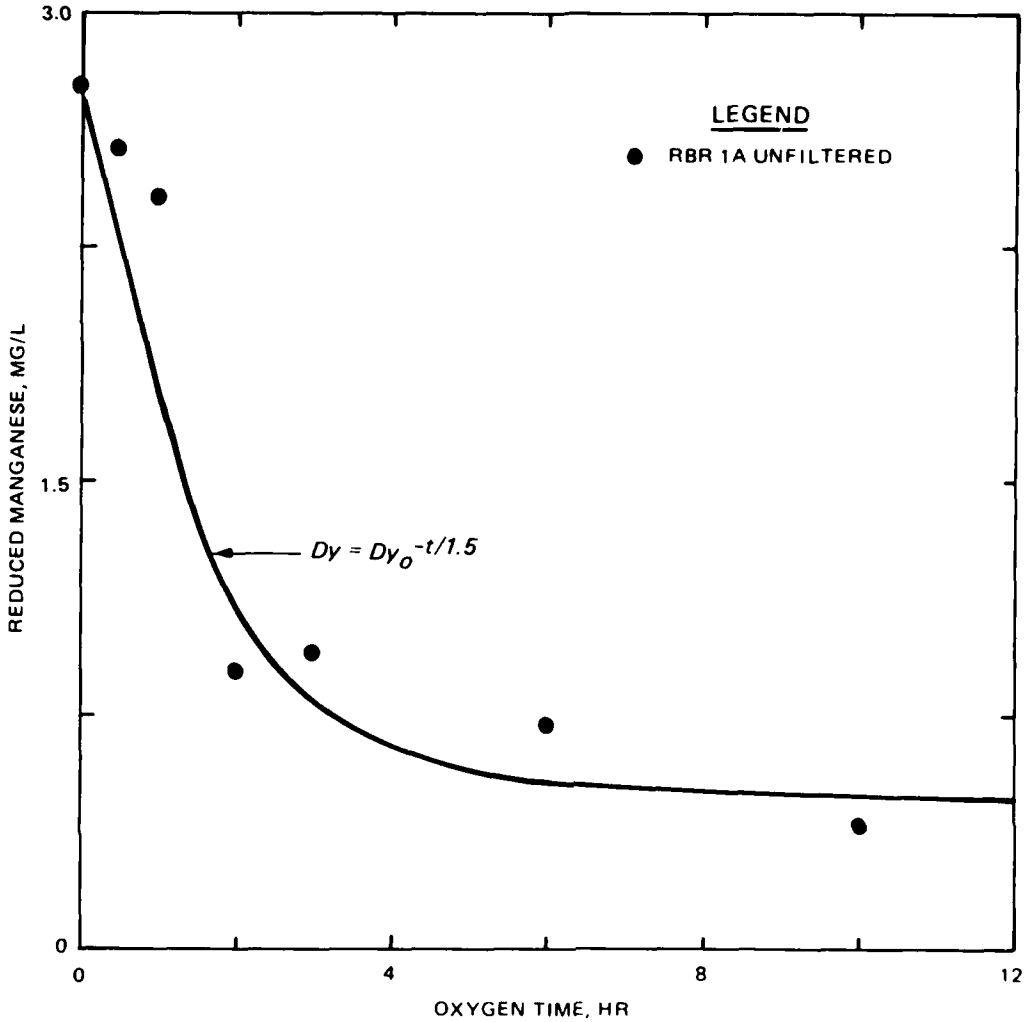

Figure VIII-7. Change in the reduced manganese concentration of an unfiltered sample as a function of duration of oxygenation of the sample. Solid curve corresponds to an exponential decay with a rate constant of 1.5 hr. Dy_o and Dy are the initial concentration and the concentration at time t, respectively

form, the experimental results do not support this conclusion at this time.

187. Similar results for a different water sample are shown in Figure VIII-8. This sample, which was unfiltered, was drawn after the lake had undergone significant mixing. The oxidation in this case appears to occur with a time constant of about 5 hr. It is not known at

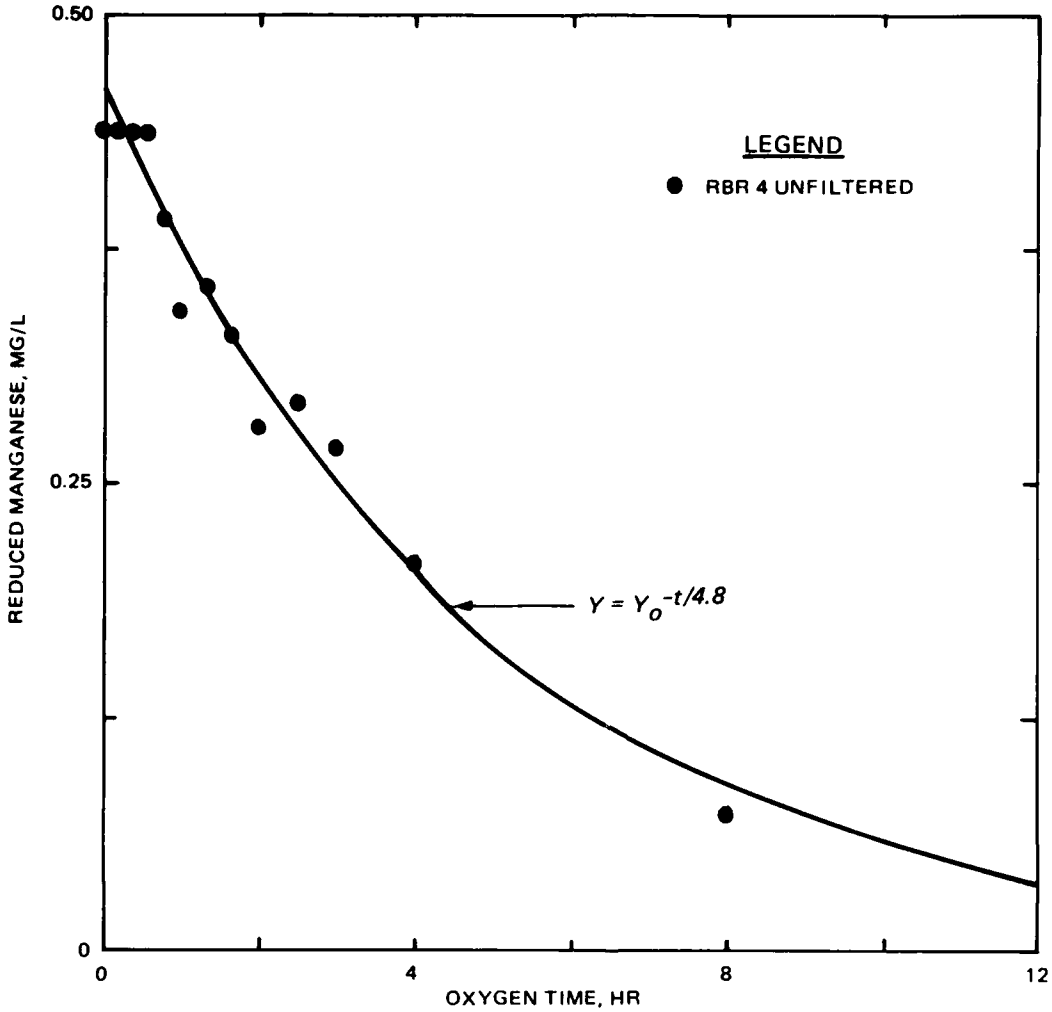

Figure VIII-8. Change in the reduced manganese concentration of an unfiltered sample as a function of duration of oxygenation of the sample. The sample was collected following fall mixing. Solid curve corresponds to an exponential decay with a rate constant of 4.8 hr

this time if this longer time constant is somehow due to the smaller manganese concentration or to a difference in the manganese environment.

188. One set of samples that had been measured by atomic absorption (AA) spectroscopy were also measured by EPR. These samples, which were drawn on 10 September 1984, were filtered through a 0.1-μ filter and acid-stabilized. The sample information and measurement results are given in Table VIII-1.

Table VIII-1

Comparison of Results Using Atomic Absorption Spectroscopy and EPR

Station	Depth m	Oxygen mg/ℓ	Manganese Concentration AA mg/ℓ	Manganese Concentration EPR mg/ℓ
80	42	0.0	3.3	2.49
120	32	0.0	2.4	2.14
140	5	1.7	0.1	0.11
150	4	0.7	0.8	0.68
160	1	24.5	<0.1	<0.08
170	10	18.2	0.6	0.57

189. In the latter four samples, the AA and EPR measurements are virtually identical, implying that all manganese in the water was reduced. The results for the first two samples imply that about 20 percent of the manganese was oxidized. This is certainly not the result that was expected, but it may be due to changes that occurred in the samples between the AA and the EPR measurements (about 2 months). Correlation of the EPR and AA results on the water will require that additional, controlled measurements to be made.

SUMMARY

190. It has been established that EPR can be used to measure reduced manganese in RBR water. The detection sensitivity is essentially the same as AA, and since AA measures all manganese present, the correlation of the two types of measurements permits monitoring of the manganese dynamics in the reservoir. Preliminary measurements of oxidation of manganese in anaerobic conditions indicated that changes occurred over a period of days at the fastest. In an oxygen-enriched environment, it was found that oxidation occurred with a time constant of several hours. Differences have been observed for manganese in

particulate matter compared to the dissolved state, but insufficient experiments have been done to permit quantifying this result.

191. While it must be emphasized that all results are preliminary, they do permit cautious optimism regarding the effects of oxygenation of the reservoir on the oxidation of manganese.

References

James, W. F., Kennedy, R. H., Schreiner, S. P., Ashby, S., and Carroll, J. H. 1985. "Water Quality Studies: Richard B. Russell and Clark Hill Lakes; First Annual Interim Report," Miscellaneous Paper EL-85-9, US Army Engineer Waterways Experiment Station, Vicksburg, Miss.

PART IX: MERCURY MOBILIZATION FROM SOIL AND ITS UPTAKE BY FISH RESULTING FROM FILLING OF RICHARD B. RUSSELL LAKE*

Introduction

192. The effects of direct mercury inputs to aquatic environments have been well documented. The tragic incident at Minimata Bay, Japan, in which 52 people died (D'Itri 1971) and the closing of the Savannah River to sport and commercial fishing from Augusta to the coast (NAS 1978), both due to waste from chloralkali plants, are only two examples. Unlike many pollutants, however, mercury is a naturally occurring element. Thus, human activities other than those resulting in direct mercury inputs may lead to elevated mercury levels in aquatic environments. Recent research suggests that the construction and filling of new impoundments may be such an activity since mercury is mobilized from inundated soil.

193. Potter, Kidd, and Stanford (1975) found that the mercury concentrations of fish from Lake Powell, 8 years after impoundment of the reservoir began, exceeded 0.5 ppm. At that time, 0.5 ppm was the upper limit for foods for human consumption recommended by the Food and Drug Administration. Though no external mercury sources could be located, Potter found biomagnification of mercury, as compared to the surrounding sandstone, by a factor of 3 in periphyton, 9 in bluegills, 31 in bass, and 43 in walleye.

194. Cox et al. (1979) and Meister, DiNunzio, and Cox (1979) found that the total mercury concentration in largemouth bass and crappie from Cedar Lake, a man-made reservoir in Illinois, often exceeded 0.5 ppm 2 years after the reservoir reached normal pool level. Fish species high on the food chain contained more mercury than species lower on the food chain. Mercury analyses of soil and sediment samples revealed that soil 20 m above the maximum pool shoreline contained the

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* Part IX was written by A. R. Abernathy, M. E. Newman, and W. D. Nicholas, Environmental Systems Engineering Department, Clemson University, Clemson, S. C.

highest mercury concentrations. Sediment from the lake bottom contained the lowest mercury concentrations, and samples taken from the shoreline area, which was submerged part of the year and dry part of the year, contained intermediate mercury concentrations. This led to the conclusion that mercury was released from the newly inundated soil by bacterial methylation and was magnified through the food chain, producing elevated mercury concentrations in predatory fish.

195. Studies by Lodenius, Seppanen, and Herranen (1982) and Lodenius, Seppanen, and Uusi-Rauva (1983) of three man-made reservoirs in Finland indicated mercury concentrations in burbot and pike that exceeded 0.5 ppm 11 and 14 years after impoundment, respectively. Lodenius concluded that mercury from the inundated soil had been mobilized into the food chain by a combination of physical mixing, microbial degradation of humic substances, methylation, and gas ebullition.

196. These and similar studies led Bodaly, Hecky, and Fudge (1984) to study mercury levels in fish prior to and following the impoundment of three reservoirs in Manitoba. Predatory fish from all three reservoirs showed elevated mercury concentrations within 2 years after impoundment and showed no evidence of decline from peak mercury levels within 5 to 8 years of impoundment. The peak mercury concentrations of fish populations were correlated with increases in lake surface area. The lake which increased most in surface area showed the greatest increase in fish mercury concentrations. Again, it was concluded that mercury was mobilized from inundated soil and magnified through the food chain.

197. Fish taken from Lakes Keowee and Jocassee (located in the same drainage basin as Richard B. Russell Lake) during the first months after these reservoirs filled contained up to 5.0 ppm total mercury (Abernathy and Cumbie 1977). This led to concern over the possible increase in mercury content of fish in the newly impounded Richard B. Russell Lake. In order to evaluate this possibility, a systematic sampling scheme was devised (Nicholas 1983) with the following objectives:

a. To determine by appropriate methods, the mercury concentrations of soil, water, and fish prior to, during, and after filling of Richard B. Russell Lake.

b. To compare the values obtained in order to test the hypothesis that natural background levels of mercury in the soil can be rapidly mobilized through microbial methylation into the waters of newly impounded reservoirs, where it can be magnified through the aquatic food chain to produce mercury concentrations in fish which exceed the current Food and Drug Administration limit of 1.0 ppm.

Materials and Methods

Sampling

198. Water, soil, and sediment samples were collected during May, June, and July 1983, before the filling of Richard B. Russell Lake (Nicholas 1983). During and after filling of the reservoir, water and soil/sediment samples were collected monthly from February through December of 1984. Three transects, corresponding to Stations 120, 180, and 190 (see Figure I-2), were sampled. Each transect consisted of five sampling stations in the following locations:

Station 1 - soil from above the flood pool elevation on the Georgia side.

Station 2 - soil and water from the newly inundated area on the Georgia side.

Station 3 - sediment and water from the Savannah River bed.

Station 4 - soil and water from the newly inundated area on the South Carolina side.

Station 5 - soil from above the flood pool elevation on the South Carolina side.

199. Soil samples were taken with an Oakfield manual soil corer. Three replicate cores were taken from each dry land station at the points of a 1-m triangle to a depth of 10 cm. The cores were transported and stored in plastic bags, frozen within 6 hr after collection, and maintained in a freezer below 0° C until the analyses could be performed. Water samples were collected with a polyvinyl chloride Van Dorn sampler from two depths in accordance with the stratification

regime at the flooded stations. One-liter samples were placed in acid-washed plastic bottles, acidified to pH less than 2.0 with concentrated nitric acid, and stored at 4° C until analysis. Sediment samples were collected from each inundated station using an Ekman dredge. Three replicate samples were taken from each haul. Samples were transported and stored in the same manner as soil samples. A sediment core was taken from transect 120, station 4, on 10 October 1984. The sediment core was transported in a clean plastic core liner, then sectioned into acid-washed glass jars, and frozen within 6 hr of collection.

200. Seventeen fish were collected from the Savannah River in September 1983, prior to the filling of Richard B. Russell Lake. The fish were taken near the South Carolina Highway 181 bridge, near the South Carolina Highway 184 bridge, and near Gregg Shoals by electroshocking or angling (Nicholas 1983). In May 1984, after the filling of the reservoir, 42 fish were taken by electroshocking from near the South Carolina Highway 72 bridge, near the South Carolina Highway 184 bridge, and from the Rocky River. In August 1984, 83 fish were taken from Beaver Dam Creek, Rocky River, and Cold Water Creek by rotenone poisoning. All fish were stored in ice until they could be frozen.

Analytical techniques

201. All analyses were performed using a modification of the cold vapor technique on a Perkin-Elmer Model 403 Atomic Absorption Spectrophotometer. The absorption cell was a glass cylinder (18 × 1.5 cm) with quartz windows cemented onto the ends. Tygon tubing was used to connect the cell in a closed system to a Cole-Parmer Masterflex rotary pump, which was operated at an air pump rate of 1.0 ℓ/min. Interchangeable tubing connectors allowed mercury vapor that had already been measured to be routed to a BOD bottle containing a scrubbing solution before the gas stream was exhausted (Hatch and Ott 1968, Nicholas 1983).

202. The 2,537-nm wavelength was used in all determinations in conjunction with a hollow cathode source lamp. The hollow cathode lamp was operated at a current of 9 mA and a monochromator slit width of 0.7 nm (Perkin-Elmer 1976, Nicholas 1983).

203. The reaction vessel was fitted with a three-way stopper with connections for the injection of the reducing reagent with a 2-ml syringe, influent air stream from the pump, and an effluent gas stream to the absorption cell. The influent gas stream was introduced into the reaction vessel through a glass tube (4-mm outside diameter) that ended 1 cm from the bottom of the vessel. When the pump was turned on, the bubbles that were produced served not only to strip the solution of mercury vapor, but also to provide mixing of the sample. Condensation of water vapor in the absorption cell was reduced by placing a 60-W light bulb 3 cm above the cell (Nicholas 1983).

Reagents and standards

204. Reagents used in the analysis were prepared as indicated in Appendix B. In all cases, A.C.S. reagent grade acids and A.C.S. certified reagent grade chemicals were used. Reagent blanks were analyzed for total mercury when contamination was suspected. Chemicals that were determined to be contaminated were discarded, and new lots were obtained. Distilled water was used for all dilutions. All glassware was washed in hot tap water in Liquinox detergent, rinsed until no trace of the detergent remained, acid-rinsed with 10-percent (V/V) nitric acid, rinsed three times with distilled water, and inverted to dry (Nicholas 1983).

Water analysis

205. Water samples were analyzed for total mercury using an acid-permanganate digestion (USEPA 1980, APHA 1981, Nicholas 1983). A standard curve of 10, 20, 40, 80, and 160 ng mercury was prepared. All samples were run in triplicate by first placing 150 ml of sample water in an acid-washed BOD bottle. Two milliliters of concentrated nitric acid, 5 ml of concentrated sulfuric acid, and 2 ml of potassium permanganate solution were then added to the sample. The sample solution was gently mixed between each addition. Two milliliters of potassium persulfate solution was added to ensure the oxidation of any phenylmercuric compounds present in the water sample. The sample solution was digested in a 90° C water bath for 30 min. Two milliliters of sodium chloride/hydroxylamine hydrochloride reagent were added

immediately before analysis. The sample was attached to the aeration apparatus. Two milliliters of stannous chloride reagent were added with a syringe, and the maximum absorbance reading was recorded. The corresponding mercury concentration was determined from the standard curve.

206. Inorganic mercury analyses of the water samples were accomplished by the differential reduction method (Magos 1971, Magos and Clarkson 1972, Nicholas 1983). A standard curve of 10, 20, 30, 40, 50 and 75 ng mercury was used. All samples were run in triplicate as follows: 200 ml of sample water was placed in an acid-washed BOD bottle, the sample was attached to the aeration apparatus, 2 ml of stannous chloride was added with a syringe, and the absorbance reading was recorded. The corresponding mercury concentration was determined from the standard curve.

Soil/sediment analysis

207. A frozen soil or sediment sample was thawed at room temperature, placed in a plastic weighing boat, and dried at 65° C for 24 hr. The sample was then reduced to a fine powder by hand grinding with a mortar and pestle. Four 1.0-g (±9 mg) aliquots of each sample were weighed into acid-washed 50-ml polypropylene centrifuge tubes. The exact weight of the soil sample was recorded to the nearest 0.1 mg. Ten milliliters of sodium hydroxide reagent were added, followed by standard additions of 0, 10, 25, and 50 ng mercury. All samples were digested in a 90° C water bath for 1 hr. After cooling the samples, 5 ml of concentrated nitric acid was added to each tube and the samples were again cooled (USEPA 1980, Nicholas 1983).

208. At the time of analysis, each sample was attached separately to the aeration apparatus and the reducing agent added with a 2-ml syringe. Stannous chloride reagent was used for the inorganic determination, and cadmium chloride-stannous chloride reagent was used for the total mercury determination. An ice bath was used to cool the reaction vessel in the total mercury determination to minimize the condensation of water vapor in the absorption cell (Magos 1971, Magos and Clarkson 1972, Nicholas 1983). The absorbance values for the four standard additions were corrected for the weight of soil in each

determination. A regression line was fitted for each set of four samples in which the x-intercept is the estimated mercury concentration in nanograms.

Fish analysis

209. Fish samples were analyzed for total mercury using an acid-permanganate digestion (APHA 1981, Nicholas 1983). A standard curve of 20, 50, and 100 ng mercury was prepared. All samples were run in triplicate by placing approximately 0.2 g of axial muscle tissue in an acid-washed BOD bottle. The exact weight of the tissue sample was recorded to the nearest 0.1 mg. Five milliliters of concentrated sulfuric acid, 3 ml of concentrated nitric acid, and 15 ml of potassium permanganate solution were added to the sample. The sample solution was gently mixed between each addition. Two milliliters of potassium persulfate solution were added to ensure the oxidation of any phenylmercuric compounds present in the fish tissue. The sample solution was digested in a 95° C water bath for 1 hr. After the sample solution had cooled, 100 ml of distilled water was added. Five milliliters of sodium chloride/hydroxylamine hydrochloride reagent was added immediately before analysis. The sample was attached to the aeration apparatus. Two milliliters of stannous chloride reagent was added with a syringe, and the absorbance reading recorded. The corresponding mercury concentration was determined from the standard curve.

210. Inorganic mercury analysis of the fish samples was accomplished using hydroxide digestion followed by the differential reduction method (Magos 1971, Magos and Clarkson 1972, Nicholas 1983). Approximately 0.2 g of axial muscle tissue was placed in an acid-washed 50-ml polypropylene centrifuge tube. The exact weight of the tissue sample was recorded to the nearest 0.1 mg. Ten milliliters of sodium hydroxide sodium chloride reagent was added, and the sample was digested in a 90° C water bath for 1 hr. After the sample had cooled, the solution was homogenized in a 15-ml tissue grinder. The solution was then diluted to 20 ml with distilled water. Four 1.0-ml aliquots of the sample solution were added to 10 ml distilled water in four 50-ml polypropylene centrifuge tubes. Four standard additions of 0, 25, 50,

and 75 ng Hg were made. Five milliliters of concentrated nitric acid was added to each aliquot, and the samples were allowed to cool.

211. At the time of analysis, each sample was attached separately to the aeration apparatus and 2 ml of stannous chloride reagent was added with a syringe. An ice bath was used to cool the reaction vessel to minimize the condensation of water vapor in the adsorption cell. The absorbance values for the four standard additions were corrected for the weight of the fish tissue in each determination. A regression line was fitted for each set of four samples in which the x-intercept is the estimated mercury concentration in nanograms.

Results

Mercury concentrations in water

212. Total mercury concentrations in the water of the Savannah River prior to the filling of Richard B. Russell Lake decreased from transect 190 to transect 120 on every occasion sampled (Nicholas 1983). This pattern was not maintained after impoundment of the reservoir. Total mercury concentrations in the hypolimnion at all three inundated stations of transect 120 rose to a peak of 0.88 µg/ℓ in May 1984. Since this time, the total mercury concentrations in the hypolimnion of stations 3 and 4 of transect 120 have remained the highest of those sampled, ranging from 0.08 to 0.26 µg/ℓ. The hypolimnion of station 4, transect 180, reached a peak total mercury concentration in June 1984 (0.14 µg/ℓ), and the hypolimnion of station 2, transect 190, reached a peak total mercury concentration in July 1984 (0.18 µg/ℓ). The time lapses between the peaks correspond to the time between inundation of the stations. The magnitudes of the peaks at transect 120 were more than three times those of the peaks at transect 180. The total mercury concentrations in the hypolimnion were generally greater than those in the epilimnion. The greatest variation between the concentrations in the hypolimnion and the epilimnion corresponded to the occurrence of the peaks (see Figures IX-1 to IX-7).

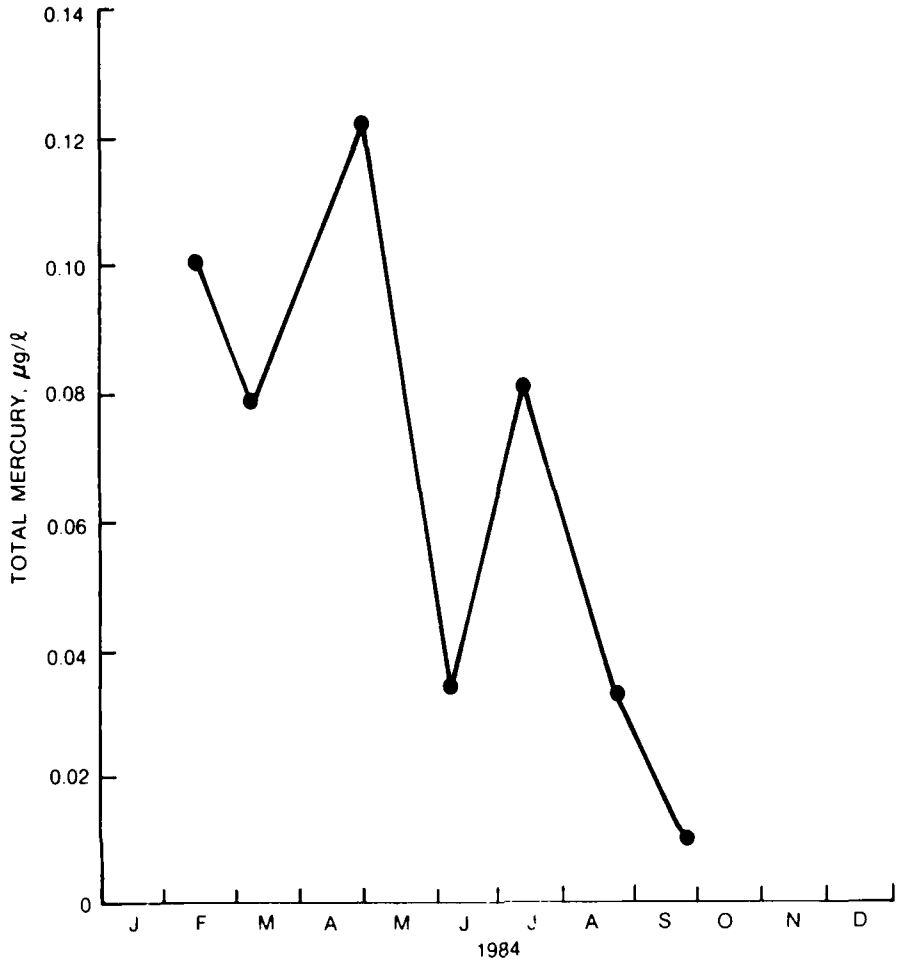

Figure IX-1. Temporal changes in epilimnetic total mercury concentration at station 3 of transect 190

213. The total mercury concentrations of the water after impoundment of Richard B. Russell Lake only approached the levels of total mercury in the Savannah River before impoundment during the peak period at transect 120. The highest total mercury concentration recorded during sampling of the Savannah River prior to impoundment was 1.120 µg/ℓ at transect 190 in July 1983. The mean total mercury concentration for all stations sampled at this time was 0.677 ± 0.402 µg/ℓ (Nicholas 1983). The highest total mercury concentration recorded in Richard B. Russell Lake after impoundment was 0.880 µg/ℓ at transect 120 in May 1984. The mean total mercury concentration for all stations

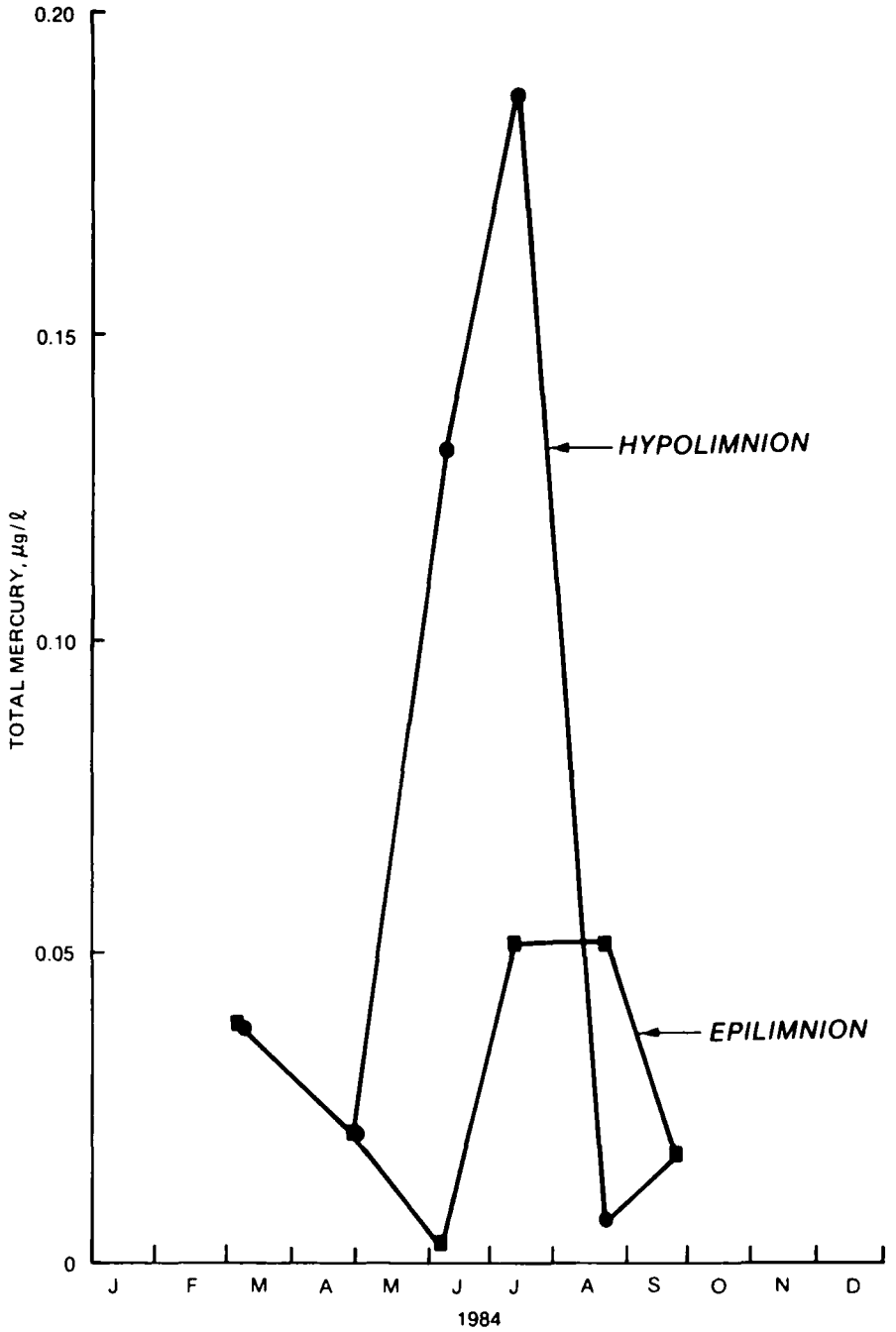

Figure IX-2. Temporal changes in epilimnetic and hypolimnetic total mercury concentrations at station 2 of transect 180

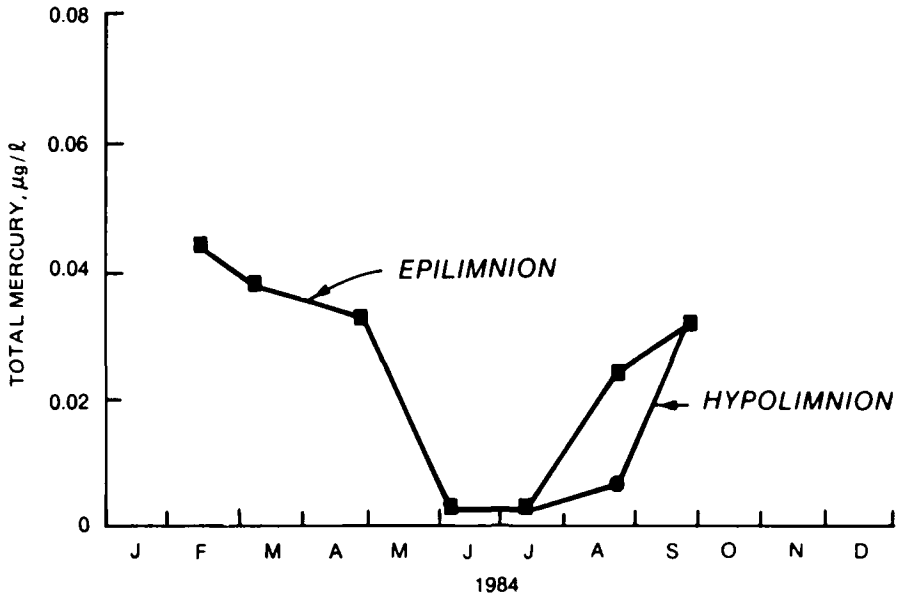

Figure IX-3. Temporal changes in epilimnetic and hypolimnetic total mercury concentrations at station 3 of transect 180

sampled at this time was 0.215 ± 0.302 µg/ℓ. USEPA reference samples run by the analysts for both the preimpoundment and postimpoundment studies had a percent difference of 9.8 and were both well within the 95-percent confidence interval established by USEPA. Variation in analysis is therefore not considered the cause of the low total mercury concentrations following impoundment.

214. The mean percent organic mercury over all stations sampled did not vary significantly over the preimpoundment or postimpoundment sampling periods. During the summer of 1983, the mean percent organic mercury over all stations sampled varied from 65.2 ± 27.6 to 75.7 ± 11.5 (Nicholas 1983). From February to October 1984, the mean percent organic mercury over all stations sampled varied from 79.0 ± 29.6 to 90.0 ± 29.0. There was approximately a 10-percent increase in mean percent organic mercury from the preimpoundment sampling period to the postimpoundment sampling period. A complete listing of mercury concentrations in Savannah River and Richard B. Russell Lake water is presented in Appendix C.

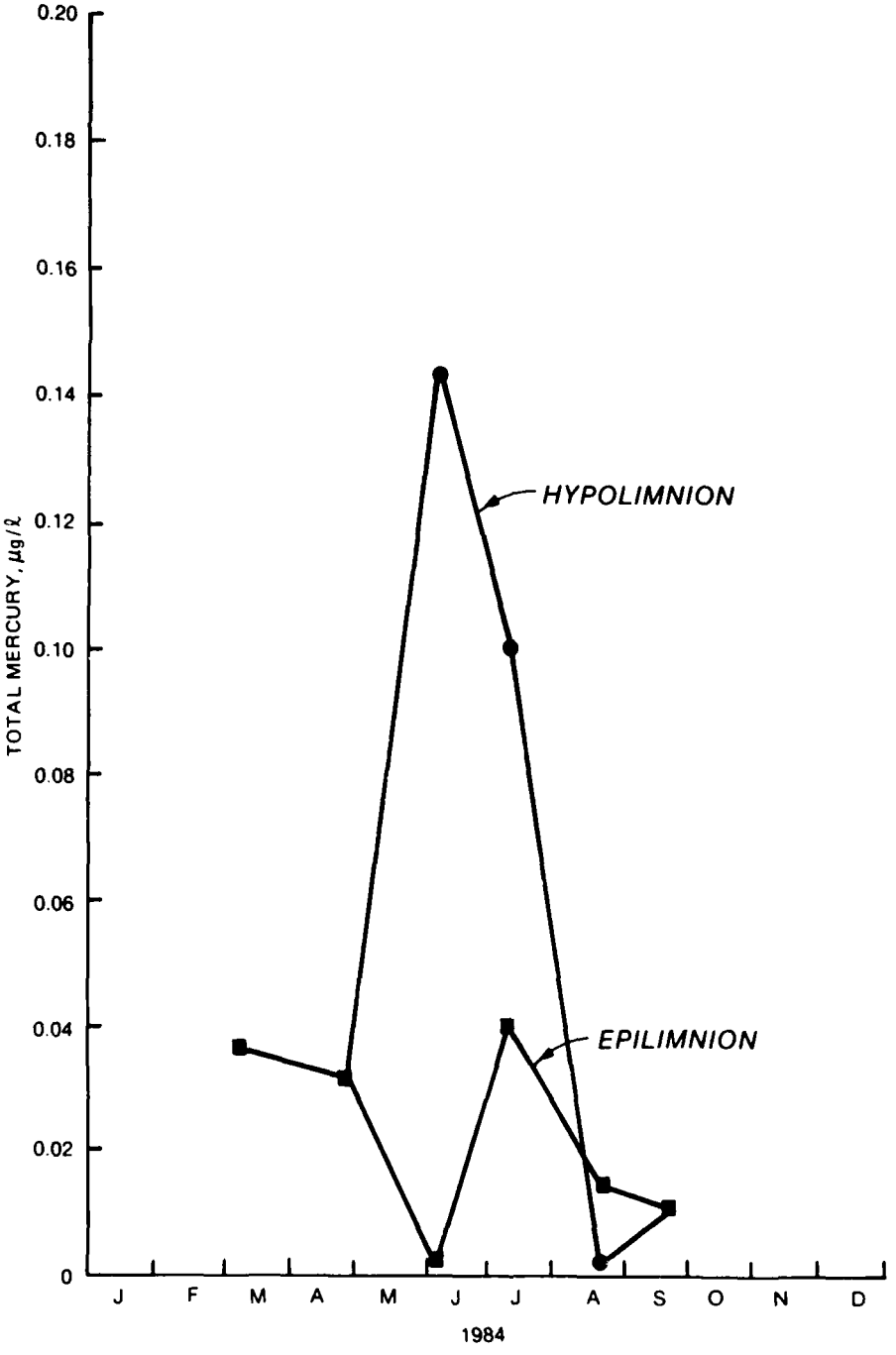

Figure IX-4. Temporal changes in epilimnetic and hypolimnetic total mercury concentrations at station 4 of transect 180

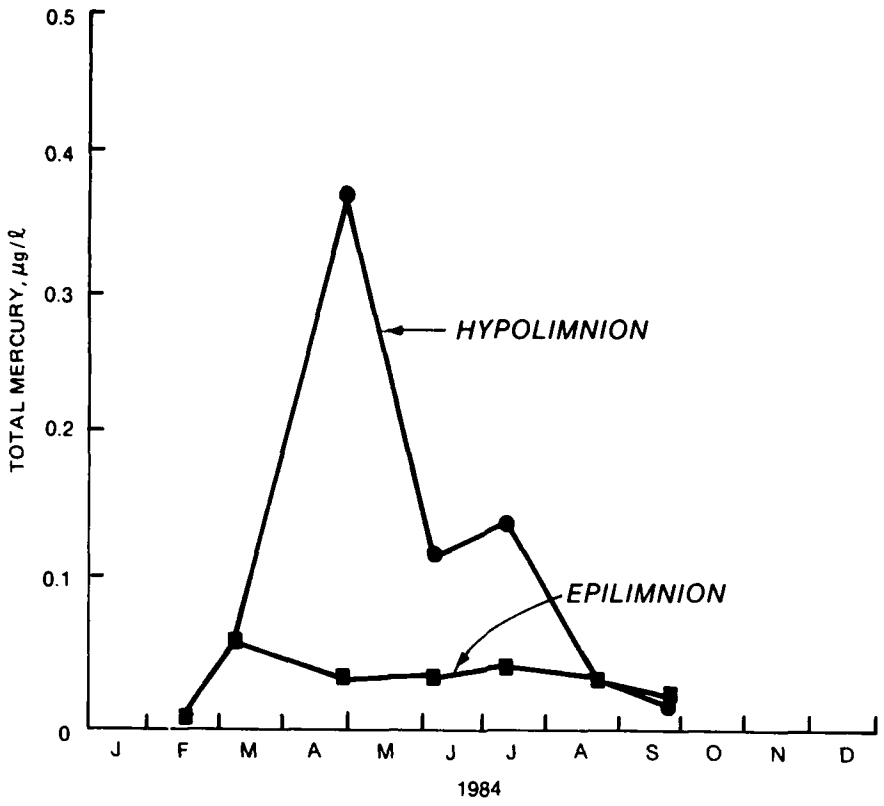

Figure IX-5. Temporal changes in epilimnetic and hypolimnetic total mercury concentrations at station 2 of transect 120

Mercury concentrations in fish

215. All fish species taken from Richard B. Russell Lake in August 1984 showed significant increases in total mercury concentration compared to values for the same species taken from the Savannah River prior to impoundment. Bass increased in total mercury concentration from a mean of 161.0 ± 136.0 ppb in September 1983 to a mean of 544.3 ± 92.4 ppb in August 1984. Sunfish increased in total mercury concentration from a mean of 80.9 ± 54.2 ppb in September 1983 to a mean of 394.8 ± 216.8 ppb in August 1984, and yellow perch increased in total mercury concentration from a mean of less than 25 ppb in September 1983 to a mean of 440.9 ± 123.6 ppb in August 1984 (see Figures IX-8 to IX-10).

216. Bass, sunfish, and yellow perch increased in total mercury concentration from May to August 1984. Though the large variations in

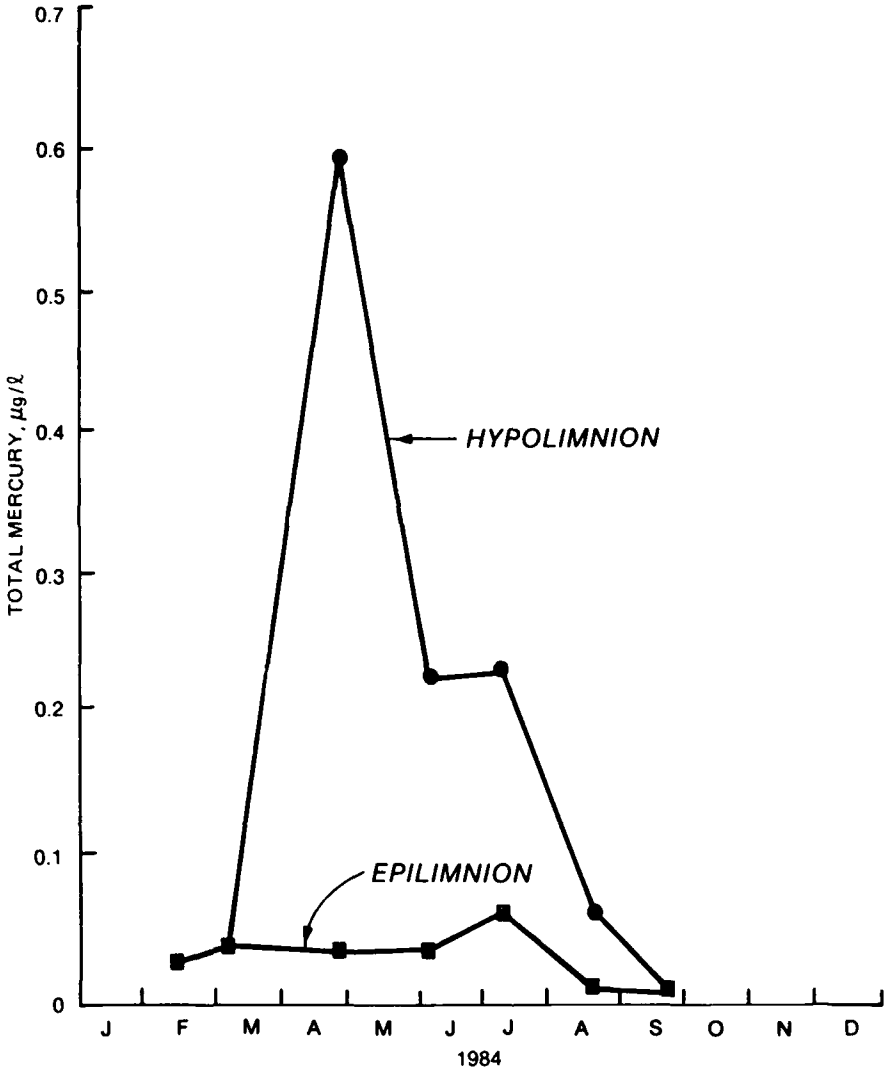

Figure IX-6. Temporal changes in epilimnetic and hypolimnetic total mercury concentrations at station 3 of transect 120

mercury content among the fish captured made the ranges overlap, a Student's t test showed the increase to be significant at the 95-percent confidence level. The mean total mercury concentration of bass collected in May 1984 was 287.9 ± 145.7 ppb. The mean total mercury concentration of sunfish collected in May 1984 was 119.0 ± 57.7 ppb, and the mean total mercury concentration of yellow perch collected in May 1984 was 104.3 ± 42.8 ppb. Catfish did not appear to vary in total

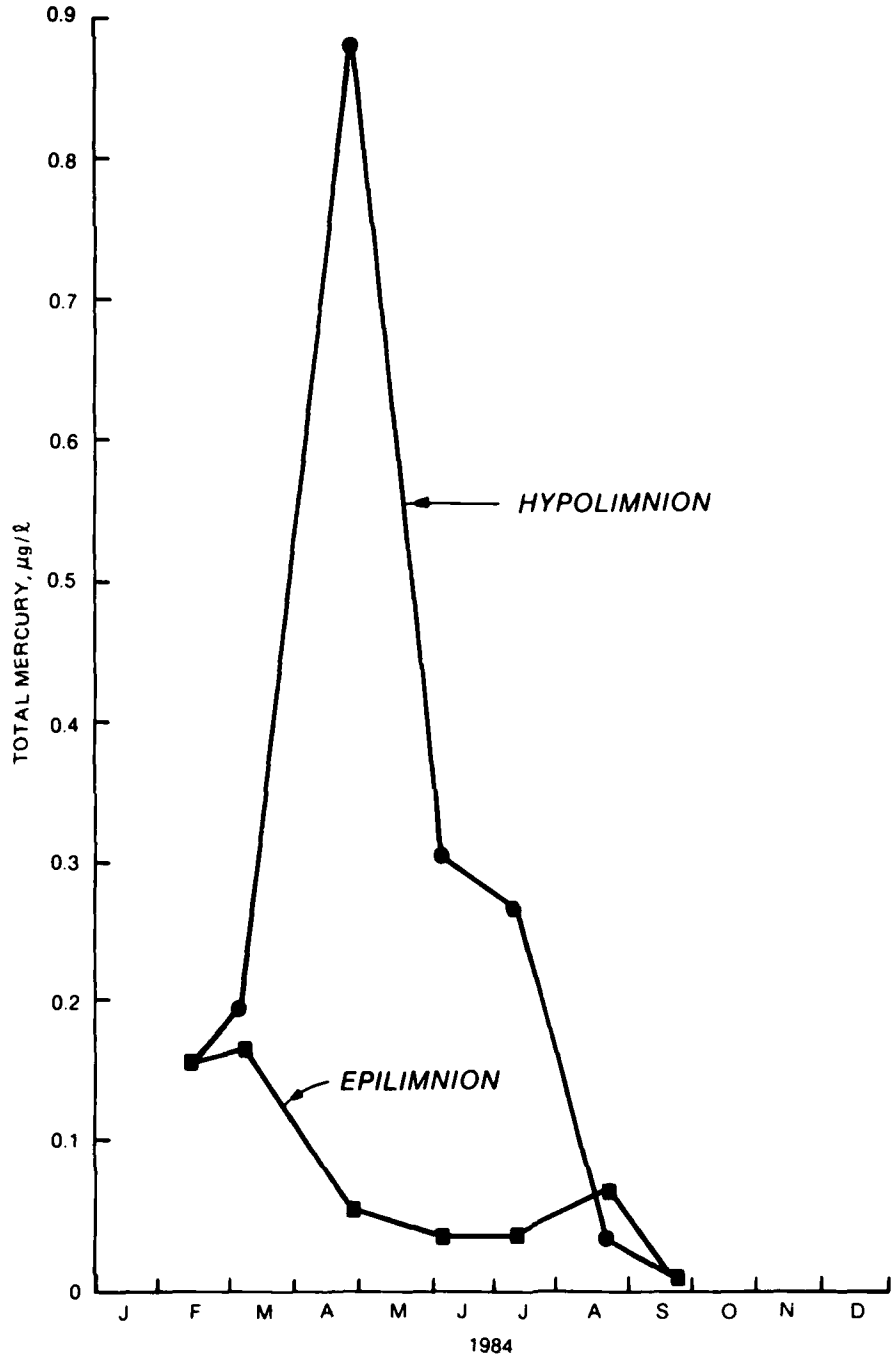

Figure IX-7. Temporal changes in epilimnetic and hypolimnetic total mercury concentrations at station 4 of transect 120

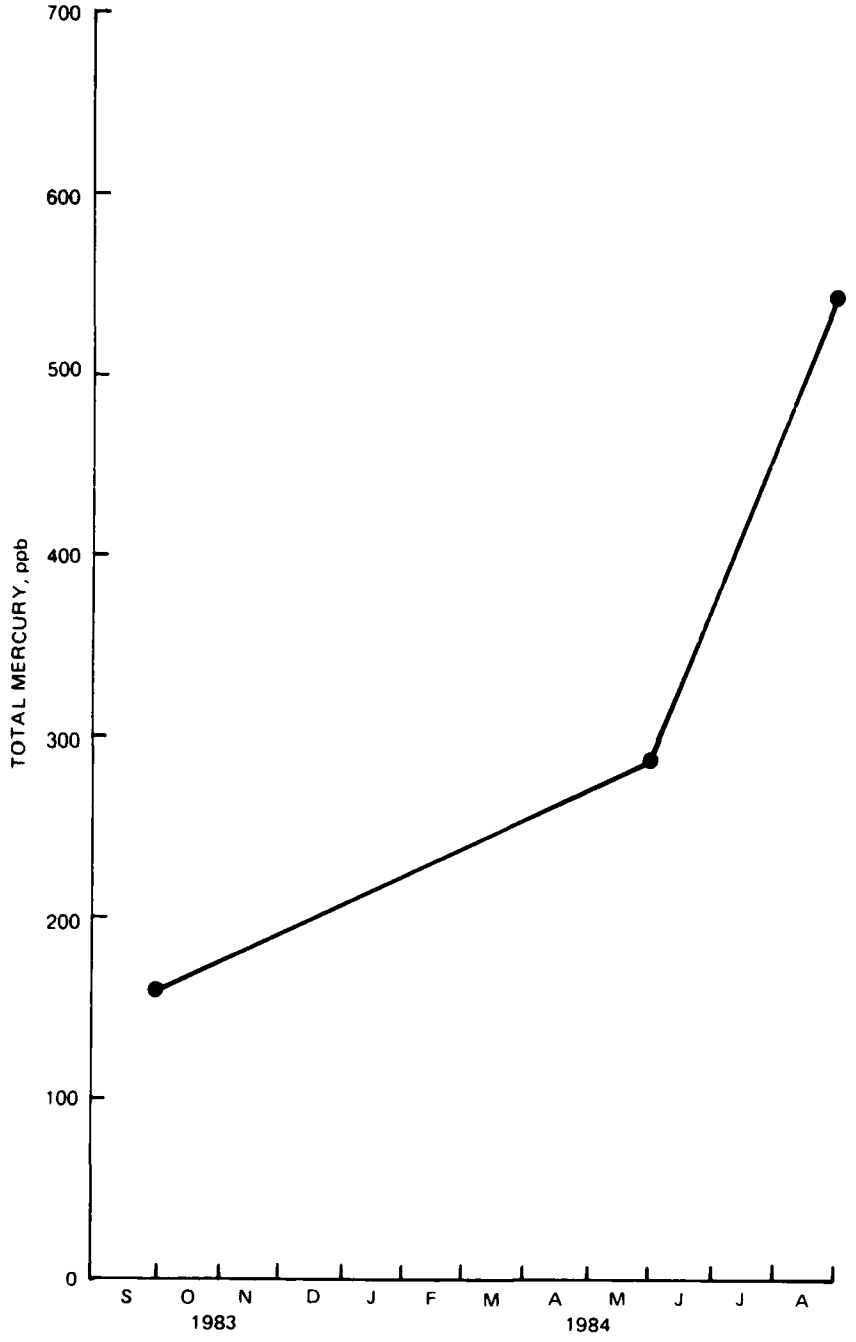

Figure IX-8. Changes in the total mercury content of bass

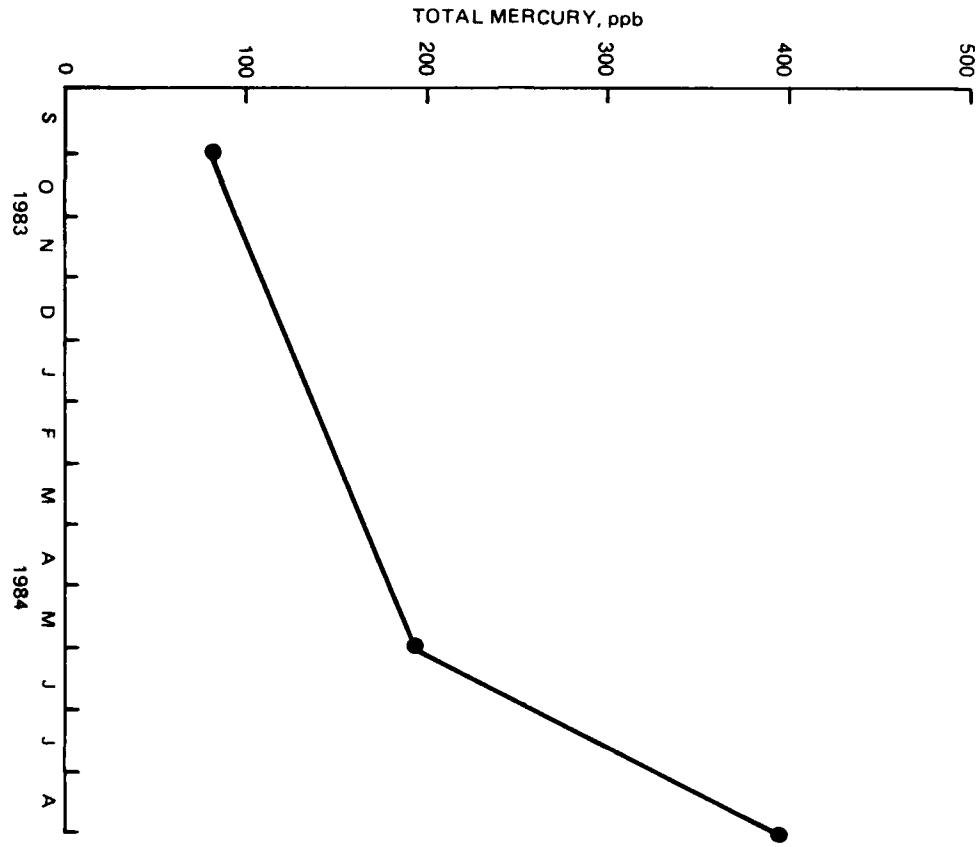

Figure IX-9. Changes in the total mercury content of sunfish

mercury concentration from May to August 1984. The total mercury concentration of the catfish collected in May was 428.0 ppb, while the mean total mercury concentration for the catfish collected in August was 433.2 ± 42.8 ppb.

217. There was little correlation between weight or length and total mercury concentration of bass, sunfish, or yellow perch collected after impoundment of Richard B. Russell Lake. Due to the limited number of fish obtained prior to impoundment of the reservoir, correlation analysis could only be performed on sunfish. This analysis also showed no correlation between weight or length and total mercury concentration.

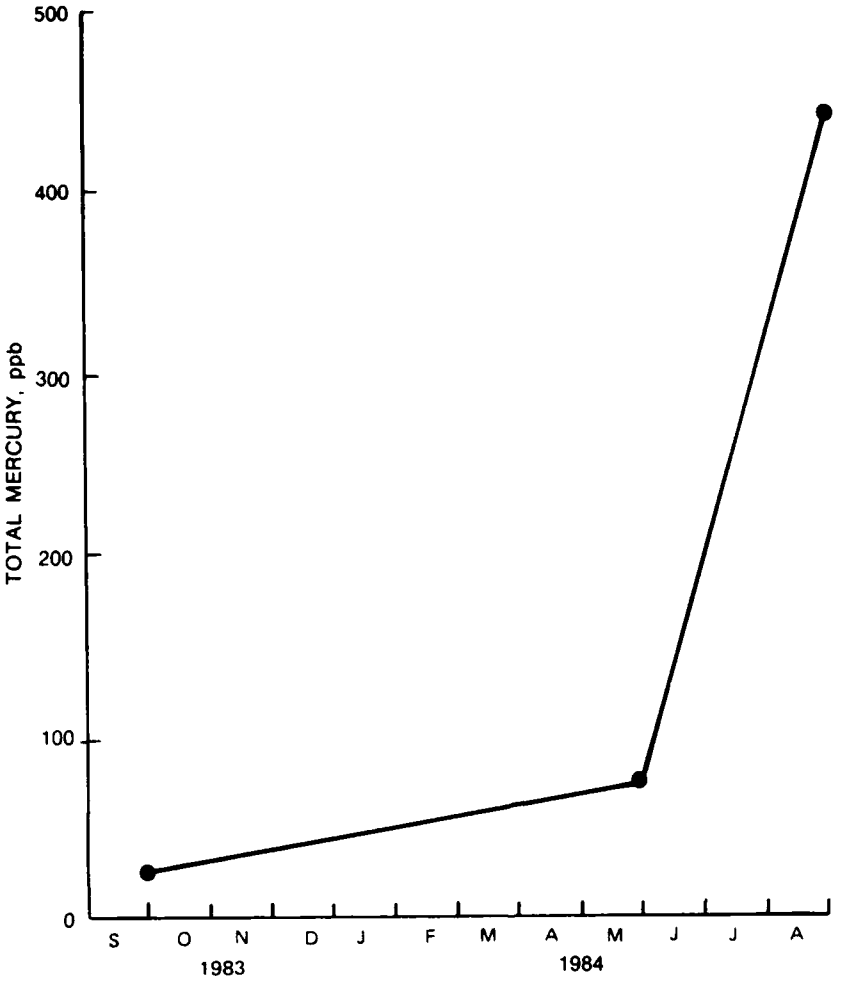

Figure IX-10. Changes in total mercury content of yellow perch

218. There were differences in the total mercury concentrations of sunfish taken from Beaver Dam Creek and those taken from Cold Water Creek. Sunfish taken from Beaver Dam Creek in August 1984 had a mean total mercury concentration of 515.5 ± 195.4 ppb, while sunfish taken from Cold Water Creek in August 1984 had a mean total mercury concentration of 247.8 ± 114.2 ppb. Though the large variation in mercury content among the fish captured made the ranges overlap, a Student's t test showed the increase to be significant at the 95-percent confidence level. A similar observation was made between the percent organic mercury content of sunfish taken from Beaver Dam Creek and those taken from

Cold Water Creek. Sunfish taken from Beaver Dam Creek in August 1984 had a mean percent organic mercury concentration of 89.9 ± 8.1, while sunfish taken from Cold Water Creek in August 1984 had a mean percent organic mercury concentration of 67.7 ± 25.8.

219. The mean percent organic mercury concentration of bass, sunfish, and yellow perch did not change significantly over time. The mean percent organic mercury concentration of bass varied from 92.0 ± 6.2 in May 1984 to 82.0 ± 12.5 in August 1984. The mean percent organic mercury concentration of sunfish varied from 96.4 ± 1.9 in June 1983 to 74.9 ± 16.3 in May 1984 and the mean percent organic mercury concentration of yellow perch varied from 52.0 ± 32.5 in May 1984 to 91.0 ± 12.2 in August 1984.

220. There was no correlation between weight or length and percent organic mercury concentration of bass, sunfish, or yellow perch collected after impoundment of Richard B. Russell Lake. Again, due to the limited number of fish obtained prior to impoundment of the reservoir, correlation analysis could only be performed for sunfish. This analysis also showed no correlation between weight or length and percent organic mercury concentration. A complete listing of fish mercury contents is presented in Appendix D.

Mercury concentrations in soil/sediment

221. Mercury analysis of soil/sediment samples is currently in progress. Sufficient data have not yet been obtained to report trends at this time.

222. Analysis of the sediment core taken on 2 October 1984 from transect 120, station 4, showed a decrease in mercury concentration with depth. The upper 0.5-cm segment contained 42.6 ppb total mercury. From 0.5 to 3.0 cm there was a total mercury concentration of 17.5 ppb, and the segment from 3.0 to 5.0 cm contained 10.6 ppb total mercury. Data for preimpoundment soil mercury content are reported in Appendix E.

Discussion

Mercury concentrations in water

223. The consistently higher total mercury concentrations observed at transect 190 before impoundment of Richard B. Russell Lake indicate that a primary source of mercury to the segment of the Savannah River under study was Lake Hartwell. Once impoundment began, mercury entering the reservoir from Lake Hartwell was diluted by water from other tributaries to the reservoir. This could be the reason mercury concentrations in the waters of the reservoir were less than the mercury concentrations of the Savannah River before impoundment.

224. The magnitude of the peaks in total mercury concentration which occurred after impoundment is correlated to the increase in lake surface area at the transects sampled. Transect 120, on that part of the reservoir with the greatest increase in surface area, increased most in total mercury concentration. The greater the increase in lake surface area, the greater the surface area of newly inundated soil. The 10-percent increase in organic mercury after impoundment could indicate a change in the primary source of mercury from Lake Hartwell effluent to mobilization of mercury from the newly inundated soil by bacterially mediated methylation. Mercury being mobilized from inundated soil by bacterial methylation or gas ebullition would primarily enter the hypolimnion. This mercury input to the hypolimnion, coupled with the lack of mixing between hypolimnetic and epilimnetic waters, could have caused the observed accumulation of mercury in the hypolimnion.

225. The total mercury concentration in the hypolimnion at transect 120 decreased sharply after anoxic conditions developed. It has been well documented (Fagerstrom and Jernelov 1972; Lodenius, Seppanen, and Herranen 1982) that mercury binds strongly with sulfide. The presence of sulfide in the anoxic waters may have caused precipitation of mercury as insoluble mercuric sulfide. The high mercury concentration in the top 0.5 cm of the sediment core taken in October at transect 120 supports this observation. It is, therefore, possible that after fall overturn, sulfide in the anoxic waters of the hypolimnion may be

oxidized, releasing mercury bound to it and resulting in a second peak in the total mercury concentration of the water column.

Mercury concentrations in fish

226. The axial muscle tissue of bass, sunfish, and yellow perch showed significant increases in mercury concentrations within a year after the impoundment of Richard B. Russell Lake began. Previous studies (Potter, Kidd, and Stanford 1975; Bodaly, Hecky, and Fudge 1984) indicate that this increase in mercury concentration could continue for 5 to 8 years. The magnitude of peak mercury concentrations and the duration of the elevation in fish mercury levels are dependent on several biotic and abiotic factors. These factors include fish diet, growth and metabolic rate, water temperature, calcium concentration, chlorine concentration, alkalinity, hardness and pH, lake productivity, and the ratio of drainage area to lake volume (Wren and MacCrimmon 1983). As these factors are different for each lake, the maximum fish mercury levels and the duration of elevated mercury levels will be different for each lake. However, it has been observed that the mercury concentrations reached are usually higher and the duration of elevated mercury levels longer, in cold, acidic oligotrophic lakes (D'Itri, Annett, and Fast 1971; Lodenius, Seppanen, and Herranen 1982). Since the waters of Richard B. Russell Lake are warm, productive, and of neutral pH, it is expected that mercury concentrations in fish will not reach the levels observed in Lakes Keowee and Jocassee or remain elevated for as long as those reservoirs. It is possible, however, that within the next year, the total mercury concentrations in pan-sized fish will exceed the Food and Drug Administration's recommended limit for consumption (1.0 ppm).

227. Though many studies have found both total and organic mercury concentrations to be related to fish size (Potter, Kidd, and Stanford 1975; Abernathy and Cumbie 1977; Cox et al. 1979; Meister, DiNunzio, and Cox 1979), these studies were performed on fish from lakes at least 2 years old. Bodaly, Hecky, and Fudge (1984) found no correlation between fish size and mercury concentrations prior to or within 2 years after impoundment of the lakes affected by the Churchill River

diversion. The fish of the Savannah River were subjected to a constantly changing environment in the flowing water of the river, and the current fish population of the reservoir has been subjected to changing conditions within their life spans. These changes in environment have led to changes in the rate of mercury accumulation and elimination by the fish. It is expected that once the fish have been exposed to the more stable conditions of the reservoir for 2 or more years, a relationship between fish size and mercury concentration will develop.

Summary

228. Total and inorganic mercury concentrations for soil, water, and fish were determined prior to, during, and after the impoundment of Richard B. Russell Lake. Due to dilution of Lake Hartwell effluent by other tributaries to the reservoir, total mercury concentrations in the water of the Savannah River before impoundment were higher than those in the reservoir. The percent organic mercury in water increased by approximately 10 percent from the preimpoundment sampling period to the postimpoundment sampling period. This was probably due to a change in the primary source of mercury from Lake Hartwell effluent to mobilization of mercury from newly inundated soil. Mercury mobilization from the soil into the hypolimnion resulted in higher concentrations of mercury in the hypolimnion. Peak mercury values in the hypolimnion were reached just prior to the development of anoxic conditions in this region. Analysis of a sediment core taken during the existence of anoxic conditions indicated that mercury was precipitated as mercuric sulfide once anoxic conditions developed. It is expected that this mercury may again be mobilized after oxygen is introduced at fall overturn.

229. Total mercury concentrations in bass, sunfish, and yellow perch increased significantly from September 1983 to August 1984. Other studies indicate this increase in fish mercury concentrations could continue for as long as 5 years. Though the peak mercury concentrations nor the duration of elevated mercury levels can be accurately predicted, neither is expected to be as great as observed in Lake Keowee or

Jocassee, due to the higher temperature and productivity of Richard B. Russell Lake. However, total mercury concentrations in pan-sized fish may exceed the Food and Drug Administration's recommended limit for human consumption within the next year.

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PART X: EVALUATION OF LONGITUDINAL GRADIENTS IN THE QUALITY OF ACCUMULATED SEDIMENTS IN CLARKS HILL LAKE*

Introduction

230. The fate of sediments transported and deposited in reservoirs is a function of hydrology, reservoir mixing, physical and biological processes, and the physical and chemical composition of the influent sediments. Hydrologic flow patterns contribute to longitudinal gradients in reservoirs, which results in the establishment of three distinct zones: (a) a riverine zone, (b) a transition zone, and (c) a lacustrine zone (Thornton et al. 1981). The unique physical, chemical, and ecological properties of these zones contribute to the quantity and quality of the deposited sediments.

231. Longitudinally decreasing inflow velocities result in sorting during sediment deposition, and advective forces within the reservoir further contribute to this sorting process resulting in a pattern of sediment distribution based on changes in energy. Different energy environments also exist in other areas of the reservoir due to secondary tributary influences, wind fetch, and hypolimnetic withdrawal zones at hydropower reservoirs. These different energy environments result in a distribution of the sediments into zones of erosion, transportation, and accumulation (Hakanson 1977). Hakanson defines zones of erosion as areas where deposition of fine materials (i.e., <6 μ) does not occur. Zones of transportation prevail where fine materials are deposited discontinuously, and zones of accumulation exist where fine materials are deposited continuously. Zones of erosion and transportation can be associated with high-energy environments and would exist near inflows and in the littoral zone. Zones of accumulation can be associated with low-energy environments and would occur in the deeper portion of the lake or in areas unaffected by hydrologic factors.

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* Part X was written by Steven Ashby, Environmental Laboratory, US Army Engineer Waterways Experiment Station, Vicksburg, Miss.

232. While areal sediment distribution is important in lake bottom dynamics, continuous sediment turnover in high-energy environments would make assessment of longitudinal gradients difficult. Examination of accumulated sediments, although subject to internal processes, should more clearly define any longitudinal gradients. The purpose of this project was to use longitudinal sediment samples along the mainstem of Clarks Hill Lake from the headwater region to the dam to examine the possibility that longitudinal gradients in sediment quality exist in accumulated sediments.

Methods and Materials

233. Since sediment range surveys indicated that sediment accumulation was greatest in the old river channel, samples were taken at the deepest point along the lake's length. Since samples were collected from a variety of energy environments, selected physical characteristics were used to identify accumulated sediments. These characteristics included a moisture contents greater than 70 percent and a particle size less than 6 μm (Hakanson and Jansson 1983). These values were obtained for surficial sediments (0 to 1 cm) that were pretreated with a wet seiving method.

234. The number of stations sampled (n) was obtained from the equation suggested by Hakanson and Jansson (1983):

$$n = 2.5 + 0.5 \sqrt{a\ F}$$

where

a = the lake area (km^2)

F = the shore development

and

$$F = \frac{lo}{\sqrt{2\ \Pi\ A}}$$

where

lo = the normalized shorelength in km

A = the total lake area in km^2

A normalized shorelength of 1930.8 km and a total area of 283 sq. km. yielded a shore development of 32. Hakanson suggests that a shore development greater than 10 is rare for lakes, however reservoirs often have a much larger shore development ratio than lakes. The number of samples required to describe this system was calculated to be 50.

235. Because emphasis was placed on the accumulated sediments of the main stem, this number was reduced to 25 on the assumption that the main stem of the reservoir was approximately half of the total area. A uniform distribution of the 25 stations from Clarks Hill Dam to Richard B. Russell Dam was selected to determine sediment quality. Stations corresponded to permanent channel markers for return sampling efforts.

236. Sediment cores were collected with a single-barrel core sampler fitted with polyethylene liners and eggshell catchers. The corer was equipped with a stabilizer fin which kept the barrel vertical as it dropped through the water. One core was collected at each station for analyses of particle size, moisture content, and total iron, manganese, phosphorus, nitrogen and organic carbon. The cores were maintained in an upright position with the overlying water still in the liner until processed in the laboratory.

237. Laboratory processing was conducted in two phases. Initially (within 36 hours of sample collection), the cores were individually sampled with the following procedure. Physical characteristics of each core (i.e., depth of sediment, presence of invertebrates and organic matter, etc.) were documented and the overlying water was siphoned off. Next the top two centimeters were sectioned off using the core-slicer employed by Gunkel et al., (1984). Three more 2-cm sections were also obtained from each core for future analyses. Each section was stored in a 6 oz. plastic bag and refrigerated until laboratory analyses could be performed.

238. The second phase of processing included sample preparations and subsampling for the various analyses (Figure X-1). The sample was

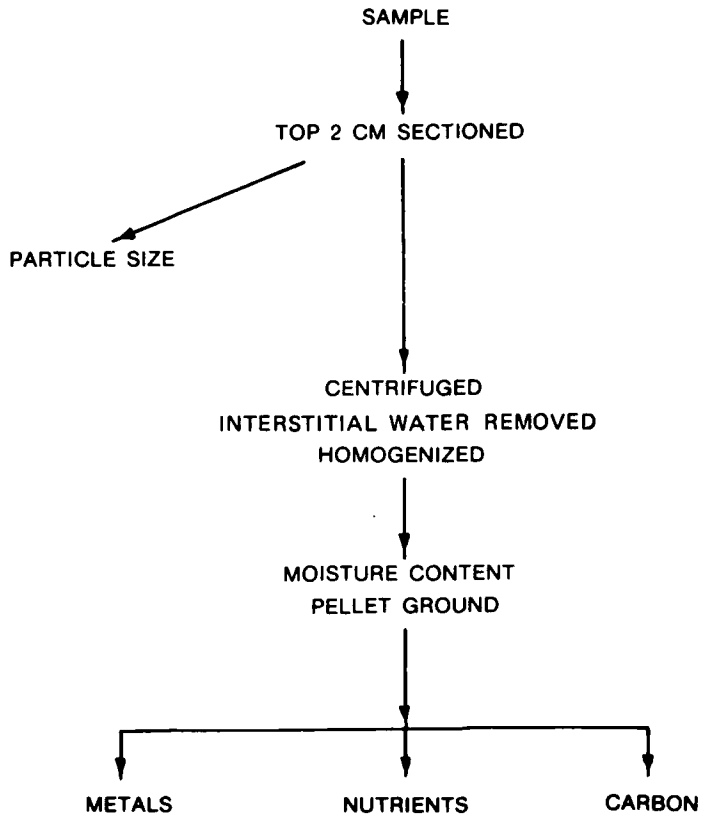

Figure X-1. Flow diagram of laboratory procedures for sediment analyses

homogenized in the bag by gentle hand melding. An aliquot was removed and dispersed in a 2-percent KCl solution for particle size analysis. The remaining sample was centrifuged at 10,000 rpm for 10 min to standardize the sediment wet moisture content. The centrifugate was poured off and the remainder of the sample hand melded for subsampling. The subsample was placed into a tared evaporating dish and weighed for moisture content analysis. The dried sample was ground with an agate mortar and pestle and then subsampled for chemical analysis.

239. Particle size analysis was performed on a Coulter Counter Model TAII equipped with a 100-μ aperture. The sample was mixed and a

10-ml aliquot was dispersed in 100 ml of 2-percent KCl. Counts per channel were converted to percent volume and summed.

240. Moisture content was determined by oven-drying approximately 7 g of wet sample (after centrifugation) at 108° C for 24 hr, then cooling in a desiccator, and reweighing. Percent moisture content was calculated as:

$$\% \text{ moisture} = \frac{\text{wet weight} - \text{dry weight}}{\text{wet weight}} \times 100$$

The drying time of 24 hr was found to be adequate based on prior studies which showed no significant changes in dry weight over multiple heating and weighing cycles. Similar results were reported by Jenkins et al. (1981).

241. The determination of total phosphorus (TP) and total nitrogen (TN) employed persulfate oxidation (Raveh and Avnimelech 1979). Approximately 0.1 g of dry sediment was digested in 5 ml of distilled water with 2 g of potassium persulfate for 1 hr at 15 psi and 105° C. This method was chosen because analyses for both parameters could be made on the same sample. Total phosphorus determinations were made by colorimetric determination, employing the molybdate reduction method. Total nitrogen determinations followed reduction to ammonium with DeVarda's Alloy and colorimetric determination employing the phenol-hypochlorite method.

242. Samples for total iron (TFE) and manganese (TMN) determinations were digested in the following manner. Approximately 0.1 g of sediment was placed in a 50-ml beaker with 50 ml of an aqua-regia acid (1:3, HNO_3:HCl) and heated uncovered in a fume hood to near dryness. After cooling, the samples were refluxed twice (heated to near dryness and the condensate retained with watch glass covers) with 3 ml of aqua-regia, then diluted to the original volume for subsequent analysis. Standards were made with aqua-regia as the matrix. Analysis was conducted on an atomic absorption spectrophotometer employing an air/acetylene flame.

243. Total organic carbon (TOC) determinations were conducted on approximately 0.1 g of sediment that was digested prior to analysis. Digestion involved phosphoric acid and potassium persulfate in sealed vials that were autoclaved at 15 psi and 105 degrees C for 1 hour. This digestion converts all organic carbon to gaseous carbon dioxide which was analyzed using a total organic carbon analyzer.

244. Replicate chemical analyses were preformed on selected samples to obtain estimates of analytical precision. Total organic carbon analyses were performed in triplicate and the mean value was reported. Precision estimates for total iron, manganese, phosphorus, and nitrogen were determined by replicate analyses of a single sample (Table X-1).

Results

245. The sample from station 25 was comprised of coarse sand, representing nonaccumulated sediment and was eliminated. Samples from the 24 remaining stations were analyzed for selected physical and chemical variables (Table X-2) and the results are presented below.

Sediment accumulation

246. In order to estimate the total depth of accumulated sediment, the presence of parent material on the bottom of the core is necessary. Parent material should be obviously different from sediment in that prior to any sedimentation the lake bottom was either covered with newly inundated terrestrial material or was old river channel. This type of lake bottom would be markedly different (i.e., covered with leaf particles, rocks, twigs, etc. or old river sand). Ten of the stations yielded cores with sediment overlying parent material. The depth of these sediments ranged from 17 to 53 cm with a mean of 31.3 cm and standard deviation of 12.5 cm. The high standard deviation may be due to basin morphometry, secondary tributary influences, and sampling methods.

Moisture content and particle size

247. Moisture content ranged from 38 to 62 percent. However, since centrifugation can remove approximately 25 percent of the moisture

Table X-1

Precision Estimates for Chemical Analyses

Variable, mg/g	$\bar{X}$ (N = 7)	Standard Deviation	Variance
TFE	61.4	2.11	3.81
TMN	2.8	0.13	0.01
TP	0.87	0.060	0.003
TN	2.02	0.094	0.008

(Hakanson 1983), a value of 25 percent was added to each sample value. The adjusted range was 63 to 87 percent. In general, moisture content increased with distance from the headwater region ($p < 0.05$). Median particle sizes ranged from 13.2 to 5.6 μ. However, greater longitudinal variability was observed for median particle size than for moisture content (Figures X-2 and X-3).

Total phosphorus and total nitrogen

248. Total phosphorus and total nitrogen values ranged from 1.02 to 0.73 mg/g and 2.5 to 1.5 mg/g, respectively. Concentrations of both variables increased with distance from the headwater region ($p < 0.05$) (Figures X-4 and X-5). Total nitrogen:total phosphorus ratios ranged from 2.0 to 2.6, with a mean of 2.3 and a standard deviation of 0.17, suggesting a uniform distribution of these variables in relation in each other.

Total organic carbon

249. Total organic carbon ranged from 12.3 to 22.1 mg/g, or 1 to 2 percent. These values are in good agreement with data from local soil surveys that indicate 1 to 3 percent values for organic material present in the soil of the watershed (USDA 1979). No significant longitudinal trends were observed (Figure X-6). A mean total organic carbon:total nitrogen value of 7.9 was observed.

Table X-2

Physical and Chemical Values

Station	Variable						
	MC*	PS**	TP †	TN †	TOC †	TFE †	TMN †
1	61.7	8.5	0.92	2.17	14.1	73.80	2.50
2	59.4	8.5	0.96	2.50	18.0	67.10	2.29
3	58.3	8.8	0.85	2.20	14.7	63.28	2.78
4	54.3	.	0.84	1.95	14.3	55.84	4.57
5	56.8	9.2	0.92	2.21	17.7	54.17	6.31
6	54.4	11.5	0.93	2.30	22.1	65.60	5.98
7	56.4	8.1	0.80	2.02	17.0	53.59	3.84
8	61.6	8.5	0.95	2.07	16.2	62.81	3.19
9	61.1	7.3	0.94	2.08	18.8	65.65	2.57
10	57.9	5.6	1.02	2.19	16.2	59.03	2.08
11	57.7	8.1	0.93	2.09	13.9	60.70	2.79
12	60.1	9.1	0.96	2.13	16.2	59.72	3.60
13	58.0	10.0	0.91	1.97	15.6	61.45	3.51
14	55.3	7.6	0.97	2.09	14.0	54.55	2.80
15	60.8	8.8	0.90	2.08	20.0	57.28	4.36
16	54.7	8.2	0.94	2.08	15.2	57.56	2.60
17	49.0	8.2	0.84	1.66	14.6	55.99	2.72
18	55.9	8.3	0.92	1.96	15.6	59.90	3.66
19	56.0	10.0	0.86	1.85	15.3	55.00	2.40
20	51.6	9.3	0.85	1.86	13.9	52.13	2.52
21	53.0	10.9	0.80	1.81	15.9	53.27	3.76
22	51.3	11.9	0.74	1.58	12.3	47.83	2.95
23	51.4	13.2	0.73	1.48	13.5	44.19	2.01
24	38.4	16.5	0.45	1.50	5.6	32.03	1.56

* Values expressed as percent.
** Values expressed in microns.
† Values expressed in mg/g.

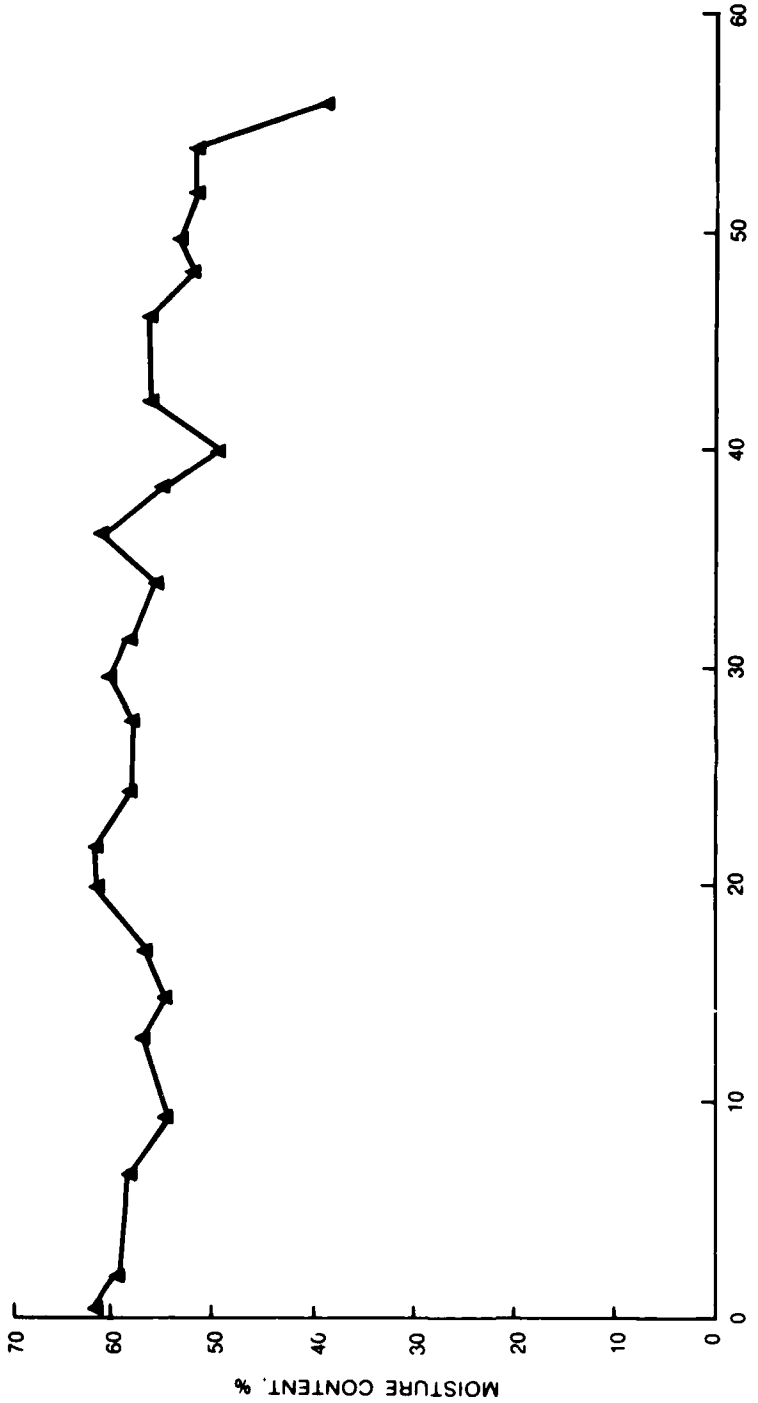

Figure X-2. Longitudinal changes in sediment moisture content

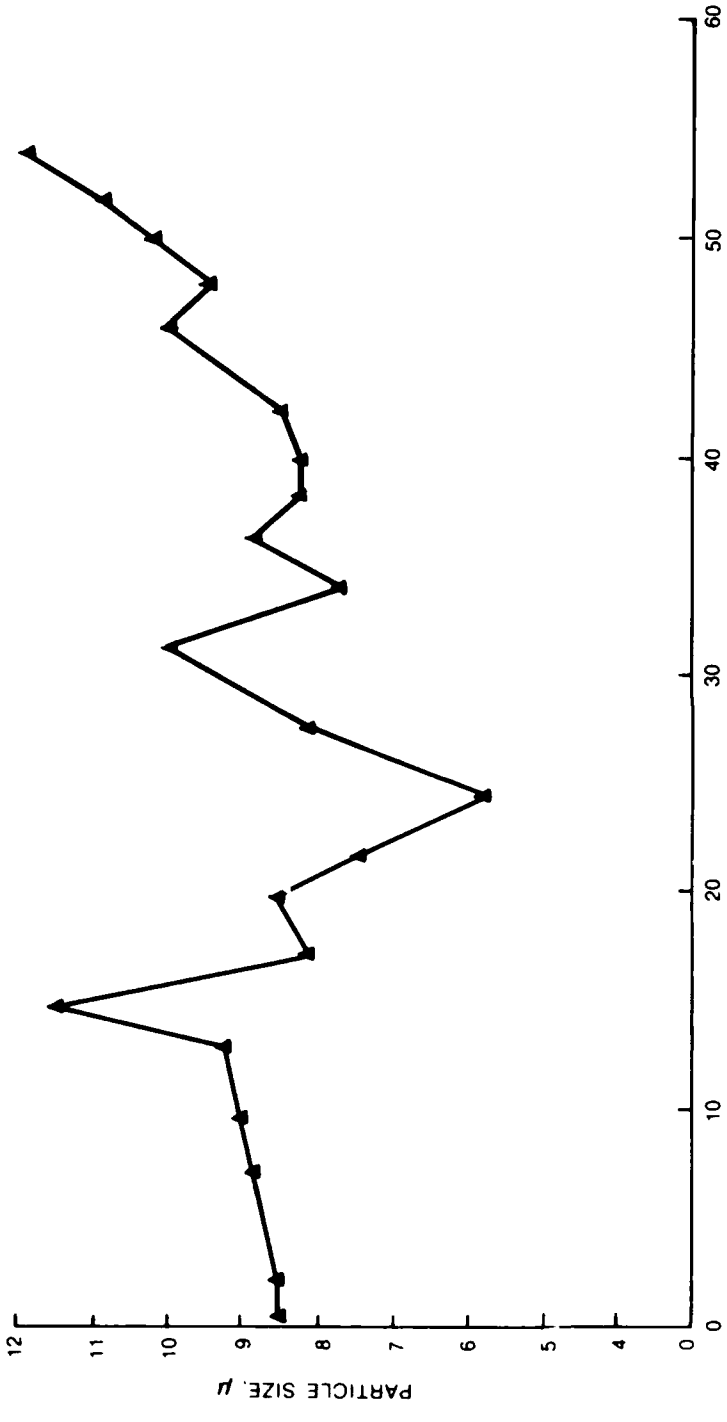

Figure X-3. Longitudinal changes in sediment particle size

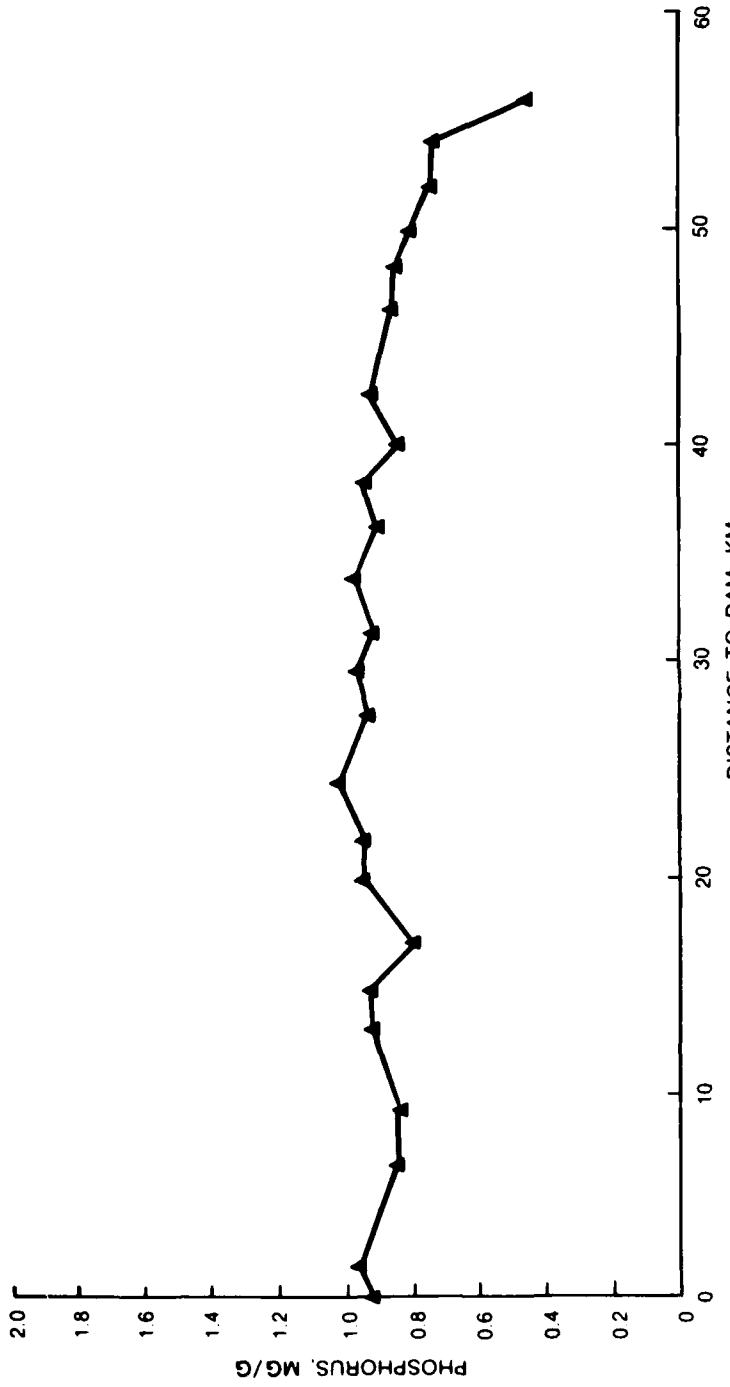

Figure X-4. Longitudinal changes in sediment total phosphorus concentration

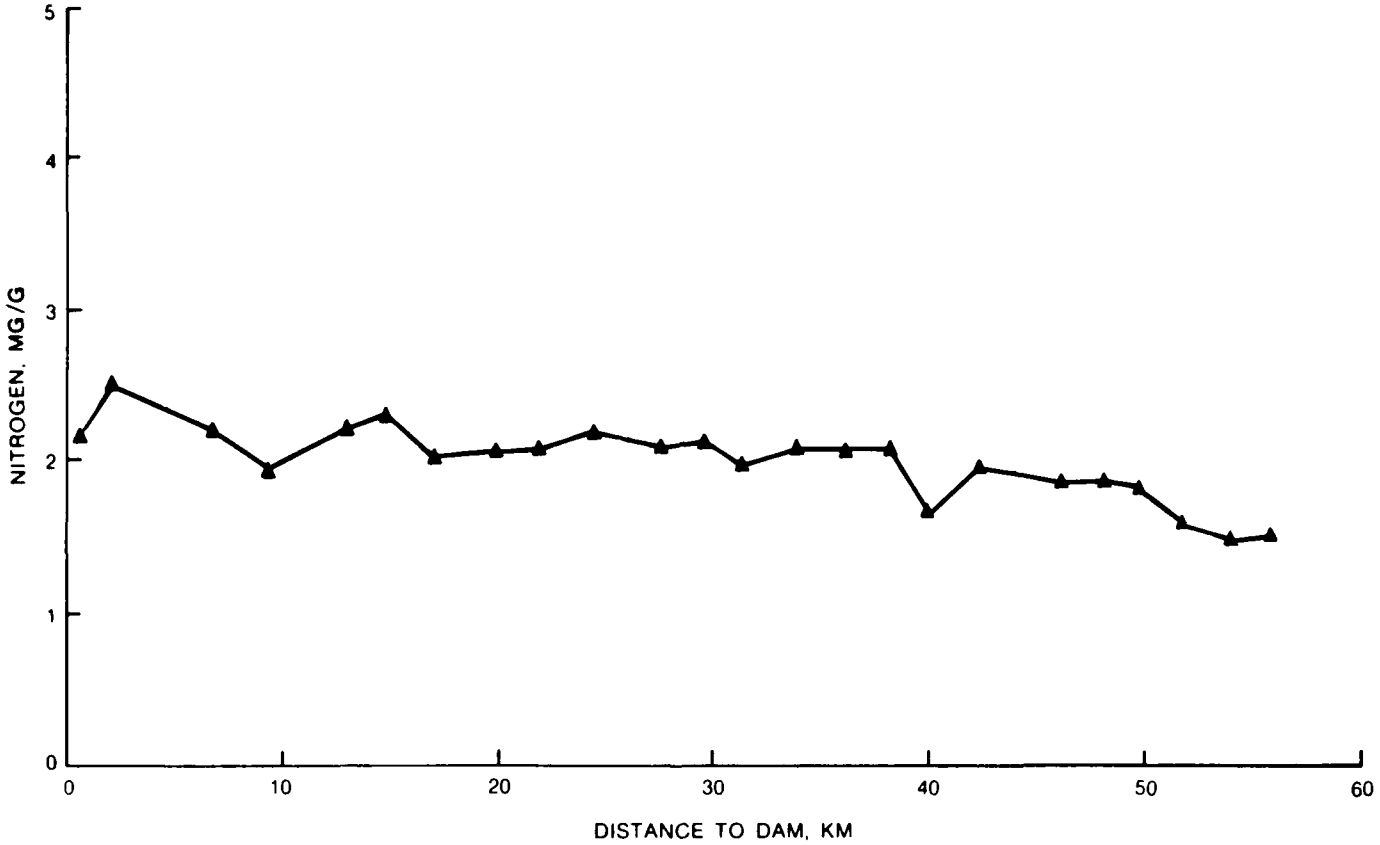

Figure X-5. Longitudinal changes in sediment total nitrogen concentration

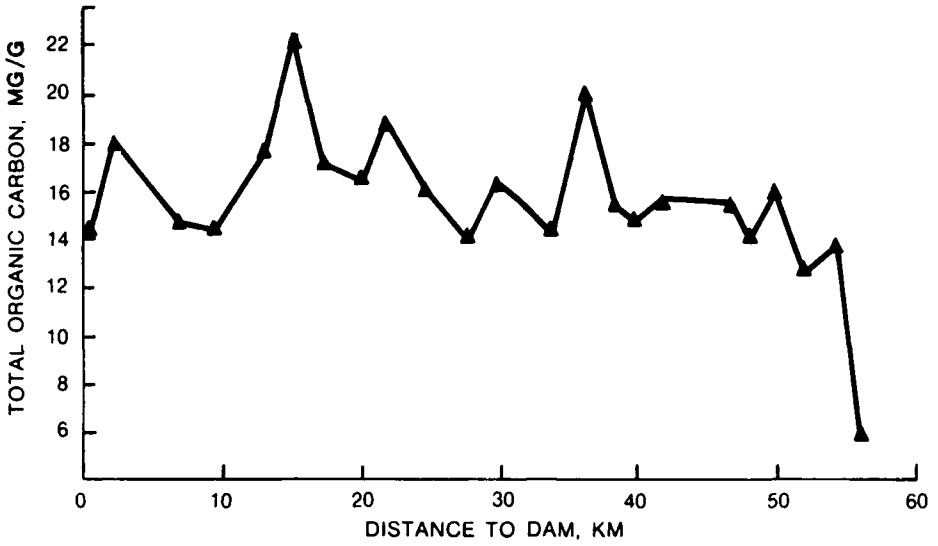

Figure X-6. Longitudinal changes in sediment total organic carbon concentration

Total iron

250. Total iron values ranged from 44.2 to 73.8 mg/g, with concentrations increasing with distance from the headwater region ($p < 0.05$). Little variation was observed longitudinally in the upstream portion of the reservoir; however, increased variability was observed in the lower region of the reservoir.

Total manganese

251. Total manganese values ranged from 2.0 to 6.3 mg/g. Longitudinal trends were not apparent ($p > 0.05$); however, a major peak was observed in the lower region of the reservoir.

Discussion

252. The quantity of sediment transported and deposited in a reservoir is a function of flow patterns, hydrologic and geologic characteristics of the watershed, physical characteristics of the influent material, and basin morphology. The quality of the deposited sediments is a function of the quality of the influent material and internal processes. While the quantity of deposited sediments is of major concern,

the quality and distribution of the accumulated sediments provide insight to the establishment of longitudinal gradients.

253. Clarks Hill Lake is a long and relatively narrow mainstem reservoir. The major inflow is the release from Richard B. Russell Dam and several secondary tributaries are located along the main stem. The quality of the major inflow is typical of reservoir releases, i.e., low in suspended sediments. This differs markedly from typical inflows into reservoirs which usually contain high sediment loads. Therefore, secondary tributaries and direct runoff into the reservoir are the major source of sediment to Clarks Hill Lake and play a major role in the deposition pattern.

254. Longitudinal gradients in sediment quality were apparent in several chemical variables. In general an overall increase in the concentrations of total phosphorus, total nitrogen and total iron was observed from the headwater region to the dam. Longitudinal gradients in total organic carbon and total manganese were not evident. However, pronounced heterogeneity in concentrations of these variables may have obscured longitudinal gradients.

255. Sediment trap data from stations 20, 30 and 40 (see Figure I-3) indicate an overall increase in concentrations of total iron, total manganese, total nitrogen, total phosphorus and total organic carbon from the headwater region to the dam* This suggests that sedimentation patterns may play a significant role in the establishment of longitudinal gradients. Thornton et al., (1981) and Kennedy, Thornton, and Gunkel (1982) have observed a relationship between longitudinal gradients in water quality and sediment transport, and deposition.

256. Patterns in sediment distribution can often be associated with water quality characteristics. Gunkel et al. (1984) suggest that similarities between patterns in water quality and sediment quality may exist in reservoirs. James et al. (1985) observed moderate spatial and

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* Personal Communication, 1986, Harry Eakin, Scientist, Environmental Laboratory, US Army Engineer Waterways Experiment Station, Vicksburg, Miss.

seasonal patterns in chemical variables in Clarks Hill Lake. Most pronounced were longitudinal gradients in total manganese concentrations during periods of thermal stratification and hypolimnetic anoxia in the lower region of the reservoir. Total nitrogen concentrations exhibited a similar pattern but were less pronounced. Longitudinal gradients in concentrations of total phosphorus, total organic carbon and total iron were not detected during the same period.

Conclusions

257. In general, longitudinal gradients in selected chemical variables were observed in accumulated sediments. Comparisons of longitudinal gradients in selected sediment variables, and those for sediment trap and water suggest a relationship to water quality and suspended material transport. Pronounced heterogeneities in concentrations of total manganese and total organic carbon suggest that seasonal water quality conditions and internal processes can impact the quality of accumulated sediments. The net result of these interactions among water quality, influent sediment, and accumulated sediment, is a cause-and-effect relationship in which the sediments serve not only as a sink for chemical variables but also as a source.

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PART XI: INTERRESERVOIR INTERACTIONS: WITHIN- AND AMONG-RESERVOIR PATTERNS IN ORGANIC MATTER PRODUCTION AND PROCESSING*

Introduction

258. The impoundment of Richard B. Russell Lake between two older reservoirs (Lake Hartwell upstream and Clarks Hill Lake downstream) on the Savannah River provides a rare opportunity to investigate the limnological and ecological aspects of interreservoir interactions and their influence on reservoir water quality. Research reported here focuses on the spatial and temporal patterns of planktonic organic matter production and processing in the Hartwell (HT)-Russell (RBR)-Clark Hill (CH) reservoir series. The objective was to document within- and among-reservoir patterns in organic matter production and processing in the reservoir series in relation to (a) the inundation of the RBR basin, (b) the subsequent aging and stabilization of the new impoundment, and (c) changes in reservoir operations. More specifically, the following research questions were addressed during this first year of study: (a) To what extent, spatially and temporally, does the leaching of nutrients and organic matter from the recently inundated basin influence planktonic production and organic matter processing in RBR and in CH downstream? (b) To what extent does terrigenous organic matter from the RBR basin become incorporated into the planktonic food chain in RBR and CH reservoirs?

Methods

Field sampling and measurements

259. During the first summer of inundation of the RBR basin, two research trips (10-12 July and 11-13 September, 1984) were made to

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characterize the spatial patterns of planktonic production and organic matter processing in the reservoir series. Nine stations along the longitudinal axis of the reservoir series were sampled: one station in HT, four stations in RBR, and four stations in CH. A near-dam station was sampled in HT as an upstream "control" not affected by the inundation of the RBR basin. Sampling stations in RBR and CH were chosen in collaboration with the US Army Engineer Waterways Experiment Station (WES) RBR Limnological Laboratory personnel and, for the most part, corresponded to established water quality monitoring stations (Table XI-1).

260. At each sampling station, vertical profiles were obtained for water temperature, conductance, pH, and dissolved oxygen with an in situ monitoring system; for photosynthetically active radiation (PAR) (400 to 700 nm), with a Li-Cor quantum meter equipped with a submersible 4-pi sensor; and for in vivo chlorophyll fluorescence (IVF), with a Turner Designs field fluorometer equipped with a large-volume, flow-through cell. Water samples were collected with an opaque hose attached to a submersible pump and stored onboard in 10-ℓ plastic containers covered to prevent exposure to direct sunlight. Small (5-ml) subsamples from selected water samples were pipetted into 20-ml vials and preserved in the field with filtered formalin for enumeration of bacterial cells. After collection, all samples were returned to the laboratory for processing. The time elapsed between sampling and return to the laboratory depended on the lake and station being sampled, but did not exceed 4 hr.

Laboratory analyses

261. Water samples for dissolved nutrient analyses [soluble reactive phosphorus, total soluble phosphorus, NO_3+NO_2-N, NH_4-N, total soluble nitrogen, and dissolved organic carbon (DOC)] were filtered through acid-rinsed glass-fiber filters (Whatman 934-AH), placed in acid-washed plastic bottles, and frozen until analysis. Unfiltered water samples for total P and total N analyses were stored similarly. All P and N analyses were conducted by standard automated techniques (Strickland and Parsons 1972; Stainton, Capel, and Armstrong 1974). Concentrations of

Table XI-1

Correspondence of Nine Sampling Stations Located Along the Longitudinal Axis of the Reservoir Series with Water Quality Monitoring Stations Established by the WES RBR Limnological Laboratory

Station	WES Station	Description
1	--	HT near-dam station
2	180	RBR uplake
3	160	RBR midlake
4	120	RBR midlake
5	100	RBR downlake
6	50	CH uplake, in RBR tailwater at buoy 151
7	--	CH midlake, at buoy 113 ~ 7 km uplake from WES Station 30
8	--	CH midlake, at buoy 65 ~ 8 km downlake from WES Station 30
9	20	CH downlake

DOC were determined by the Menzel and Vaccaro (1964) method using an Oceanography International infrared carbon analyzer.

262. Suspended particulate matter samples were collected by low-vacuum filtration (<100 mm Hg = <13 kPa) through glass-fiber filters, and duplicate or triplicate samples were analyzed for chlorophyll a (Marker, Crowther, and Gunn 1980), adenosine triphosphate (Holm-Hansen and Booth 1966), and particulate carbon and nitrogen (Sharp 1974).

263. The IVF of filtered and unfiltered water samples was determined in the laboratory using an Aminco fluorocolorimeter fitted with a blue excitation lamp (4 W m^{-2} intensity), a blue excitation filter (420 nm, CS5-60), a red emission filter (650 nm, CS2-64), and an R-136 photomultiplier to permit a correction of field IVF profiles for the variable presence of dissolved fluorescent compounds (dissolved humic substances) at different depths and stations.

Bacterial cell
counts and growth rates

264. Formalin-preserved samples obtained for bacterial cell counts were examined using the acridine orange direct count (AODC) technique as modified by Hobbie, Daley, and Jasper (1977). Subsamples were stained with acridine orange, filtered onto 25-mm Nucleopore filters (0.2-μ pore size) stained with irgalan black (Ciba-Geigy), and examined via epifluorescence microscopy at 1000X magnification.

265. To obtain an estimate of bacterial growth rates in the mixed layers of the three impoundments, a time-series experiment was conducted with mixed-layer (upper 5 m) samples from the downlake stations of HT, RBR, and CH reservoirs (i.e., from stations 1, 5, and 9) during the September sampling trip. Replicated (n = 6) 1-ℓ plastic bottles were filled with mixed-layer water from each downlake station and incubated in the dark for 2 hr. At 0, 1, and 2 hr, a 10-ml subsample was removed from each bottle and preserved with 1-ml filtered neutral formalin. The preserved subsamples were returned to the laboratory, and bacterial cells were enumerated as described above.

Estimates of relative phytoplankton productivity

266. Photosynthesis-chlorophyll-light relationships from the literature and from the results of the authors' own investigations were used to make estimates of relative phytoplankton productivity for all nine sampling stations in the reservoir series. Specifically, a classic hyperbolic photosynthesis-light relationship was assumed, a P_{max} value of 2.0 mg C fixed/mg Chl a/hr, and an I_k value of 100 $\mu E/m^2/sec$. These assumptions were based on the results of investigations of phytoplankton photosynthesis-light relationships in a variety of southeastern US impoundments (Groeger 1986). Light at specific depths was estimated assuming an average surface light intensity of 1500 $uE/m^2/sec$ and applying the vertical extinction coefficient calculated for each station from the PAR vertical profile. Chlorophyll concentrations at specific depths were calculated from the relationship of field measurements of chlorophyll IVF (corrected for the presence of background dissolved fluorescent compounds) and extracted chlorophyll measurements on samples from each station. Therefore, if PAR at a depth exceeded the I_k value, then, relative phytoplankton productivity (PPR) = Chl * 2 . If PAR at a depth was less than I_k , then relative PPR = (PAR/100) * 2 . Our assumed P_{max} value is on the low side of the observed P_{max} range, so our relative PPR values are conservative estimates.

Stable carbon isotope analyses

267. Water samples were collected from reservoir mixed layers and hypolimnia at selected stations for determination of the stable carbon isotope composition of dissolved inorganic carbon (DIC), DOC, and particulate organic carbon (POC). All water samples were processed onsite on the day of collection in preparation for stable carbon isotope analyses by isotope ratio mass spectrometry. Whole 2-ℓ water samples were preserved with 2-ml saturated $HgCl_2$ for DIC-$\delta^{13}C$ analyses. Water samples were filtered through precombusted (550° C) glass-fiber filters (Whatman 934-AH) and $HgCl_2$-preserved for DOC-$\delta^{13}C$ analysis. Particulate matter retained on the glass-fiber filters and zooplankton collected by

vertical net hauls were stored frozen. All samples were shipped to Coastal Science Laboratories, Inc., where carbon isotope ratio measurements were performed by methods described by Parker et al. (1972).

Results

Water quality and nutrient data

268. Water quality and nutrient data for the period October 1983 to December 1984 are presented and discussed extensively by James et al. (1985). Additional physical-chemical data were collected for the purpose of this study at HT, RBR, and CH reservoirs on 10-12 July and 11-13 September.

Temporal and spatial patterns in seston chemistry and phytoplankton biomass

269. Concentrations of POC, particulate organic N (PON), and algal chlorophyll a declined throughout the multiple-impoundment series between the July and September sampling dates (Figures XI-1, XI-2, and XI-3). Particulate C and N concentrations were slightly lower in HT than in RBR and CH in July, but became more equivalent in the three reservoirs by September. HT was the most transparent and the least productive of the three reservoirs as indicated by its consistently greater photic layer depth (Figure XI-4) and lower phytoplankton standing crop (as reflected by chlorophyll a concentrations, Figure XI-3) relative to those in RBR and CH, downstream. In terms of volumetric chlorophyll a concentrations, HT was about 30 percent as productive as RBR and CH in July and September (Figure XI-3).

270. Integral and mean values for photic zone POC, PON, chlorophyll a, and PPR (estimated from light and chlorophyll levels) clearly show that a marked reduction in PPR occurred between July and September in the reservoir series (Figures XI-5 and XI-6). By September, chlorophyll concentrations decreased to about one-half the July values throughout the reservoir series. The greatest relative decline in

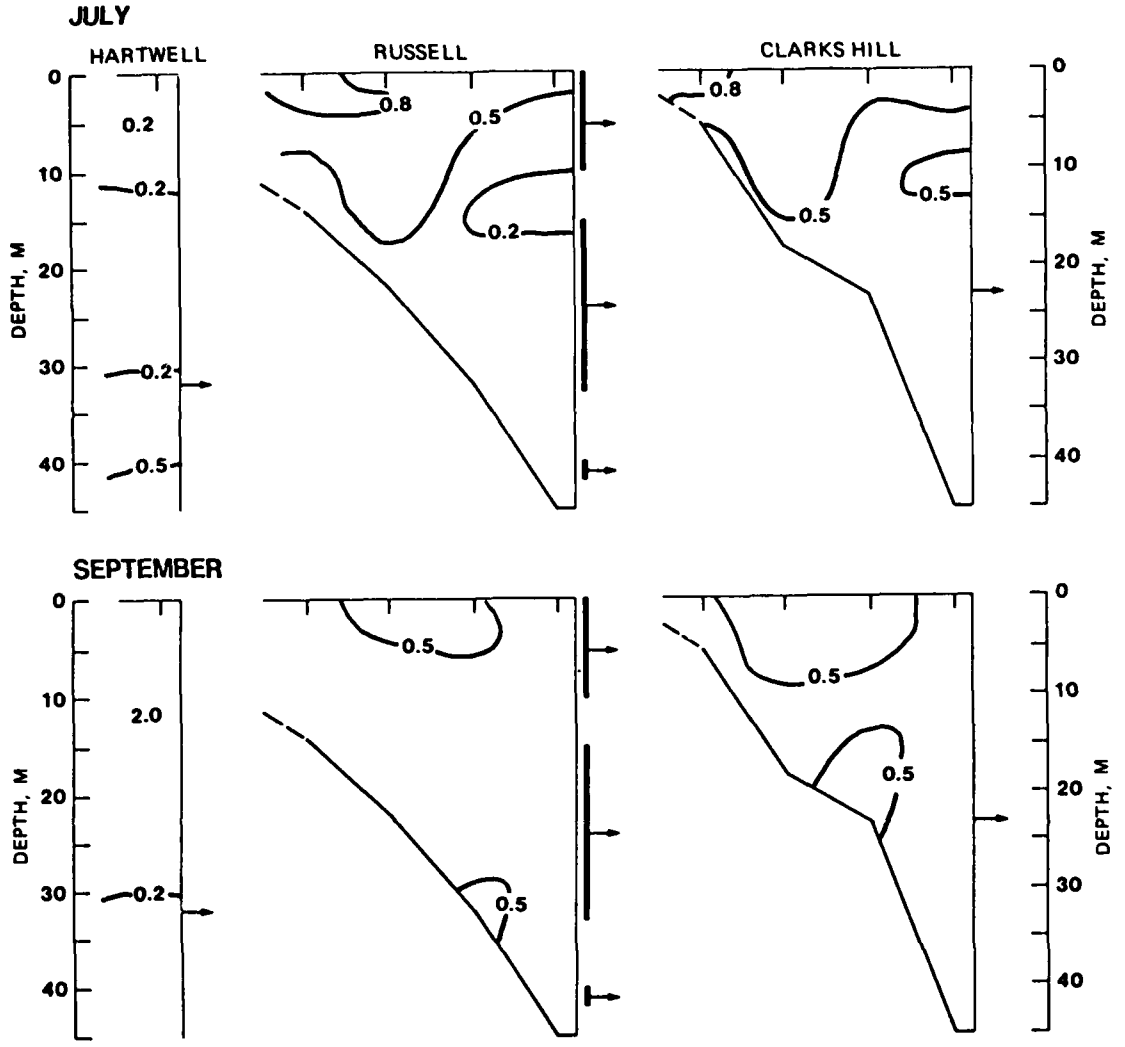

Figure XI-1. Isopleths of POC concentrations (gm/m^3) in the HT-RBR-CH reservoir series, 10-12 July and 11-13 September, 1984

integral photic zone algal biomass occurred in RBR which, by September, had a lower phytoplankton standing crop than CH (Figure XI-5).

271. Longitudinal patterns in phytoplankton biomass (as reflected by chlorophyll a concentrations) were particularly marked within and among reservoirs (Figures XI-5 and XI-6) in July, RBR exhibited a pronounced longitudinal gradient in euphotic layer chlorophyll concentrations (Figures XI-3 and XI-6). Highest chlorophyll concentrations

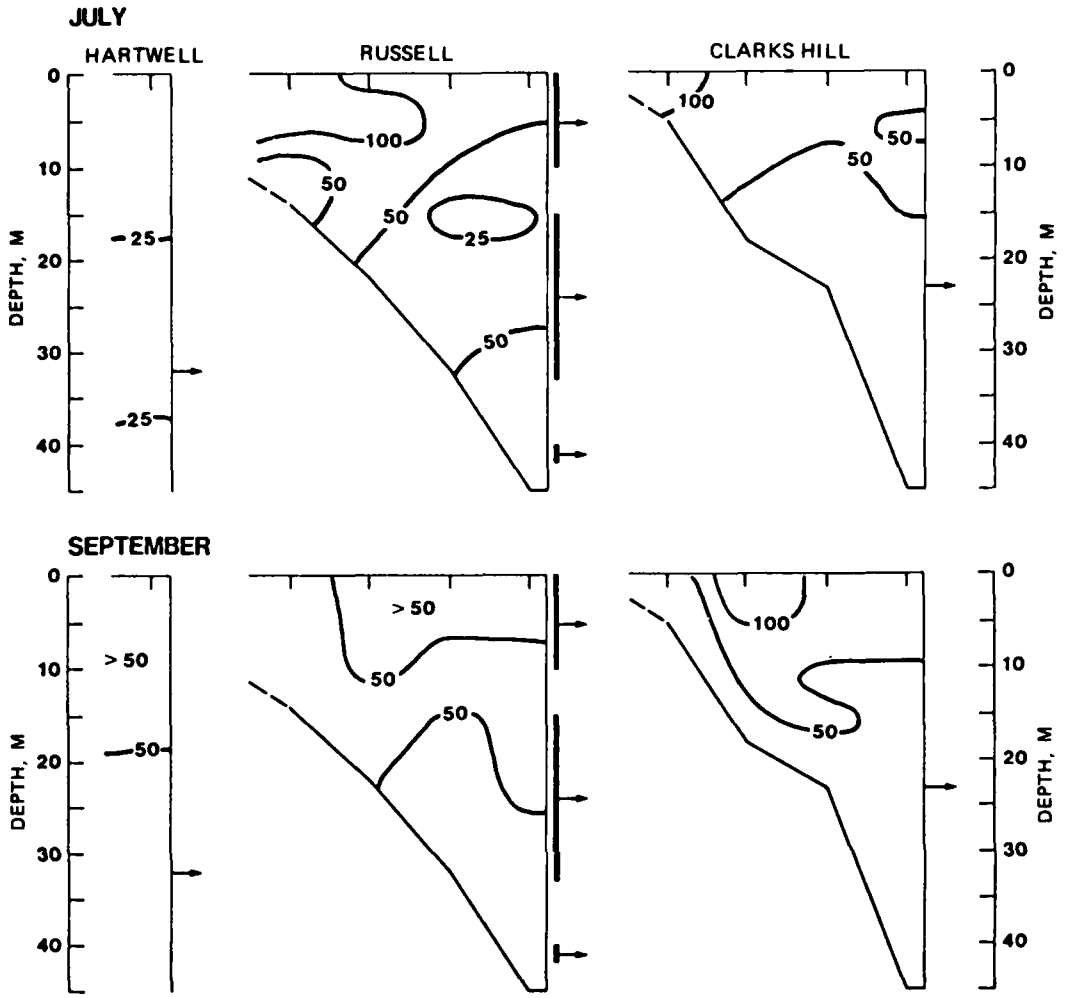

Figure XI-2. Isopleths of PON concentrations (mg/m^3) in the HT-RBR-CH reservoir series, 10-12 July and 11-13 September 1984

(~20 mg Chl *a* m^{-3}) occurred in the uplake portion of RBR and declined to 5 to 10 mg Chl *a* m^{-3} downlake. In CH, the region of highest chlorophyll concentration (~15 mg Chl *a* m^{-3}) occurred in the midlake portion of the reservoir and declined to 5 to 10 mg Chl *a* m^{-3} near the dam. Longitudinal trends within and among reservoirs were less clear in September

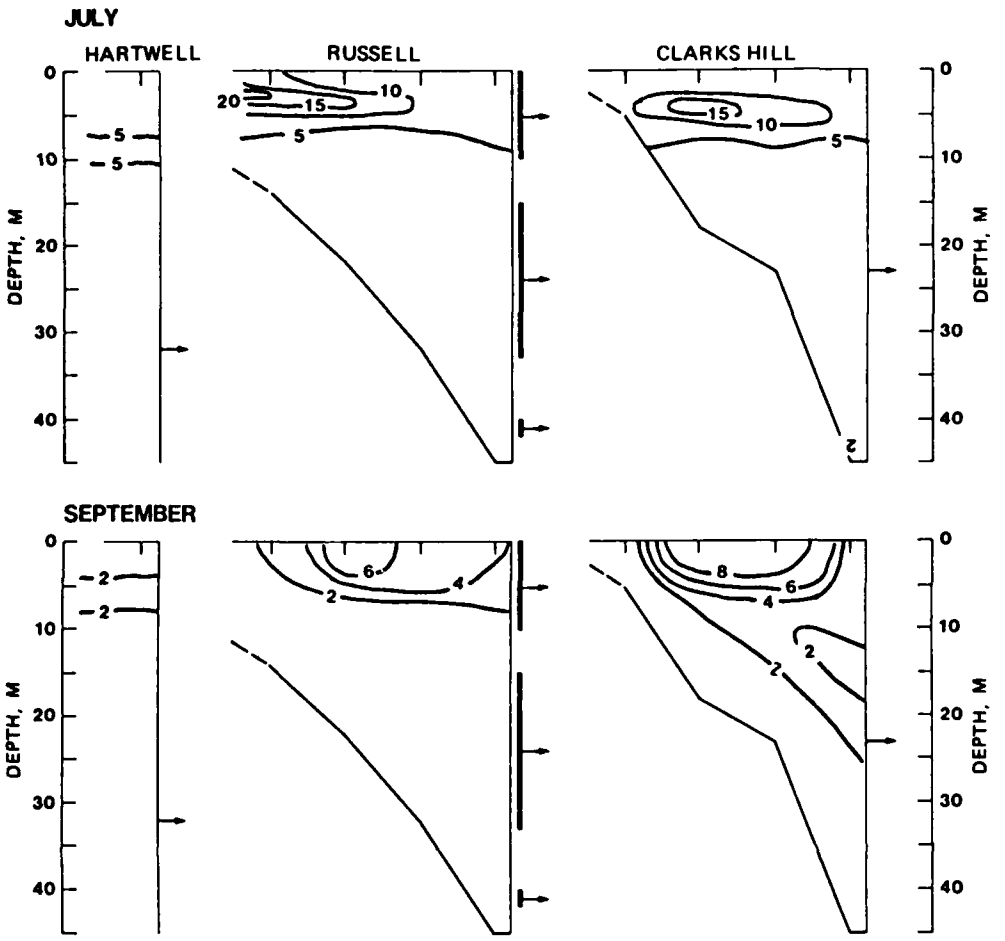

Figure XI-3. Isopleths of chlorophyll a concentrations (mg/m^3) in the Hartwell-Russell-Clarks Hill reservoir series during 10-12 July and 11-13 September 1984

as productivity levels decreased and dam discharges increased. The (PPR) decreased markedly throughout the reservoir series, but decreased to the greatest extent in RBR (Figures XI-5 and XI-6). In July, RBR was considerably more productive than both HT and CH. However, by September, RBR productivity had decreased to the point that RBR was less productive than CH, but more productive than HT.

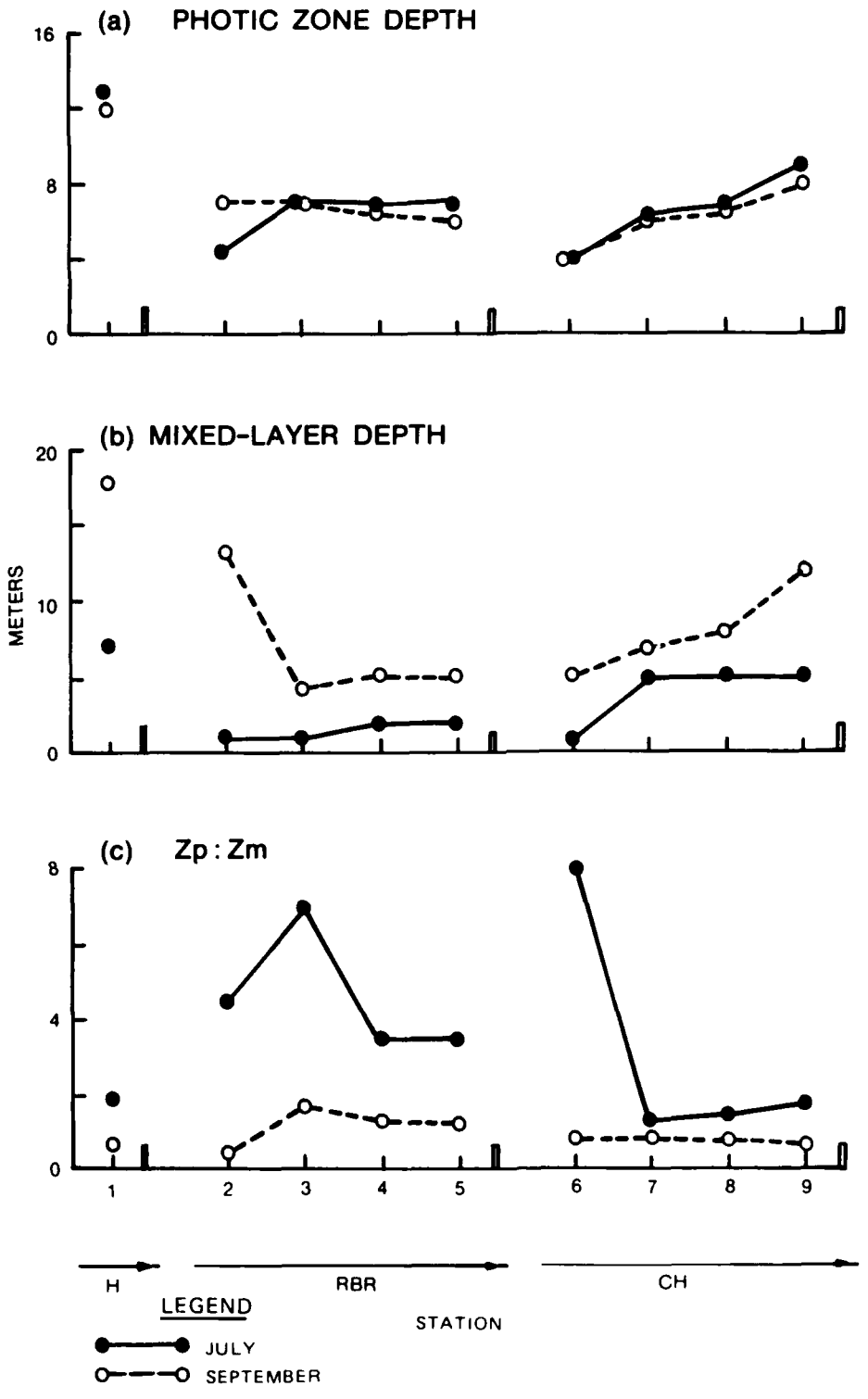

Figure XI-4. Photic zone depth (a), mixed layer depth (b), and Zp:Zm ratio (c) at nine stations along the longitudinal axis of the HT-RBR-CH reservoir series, July and September 1984

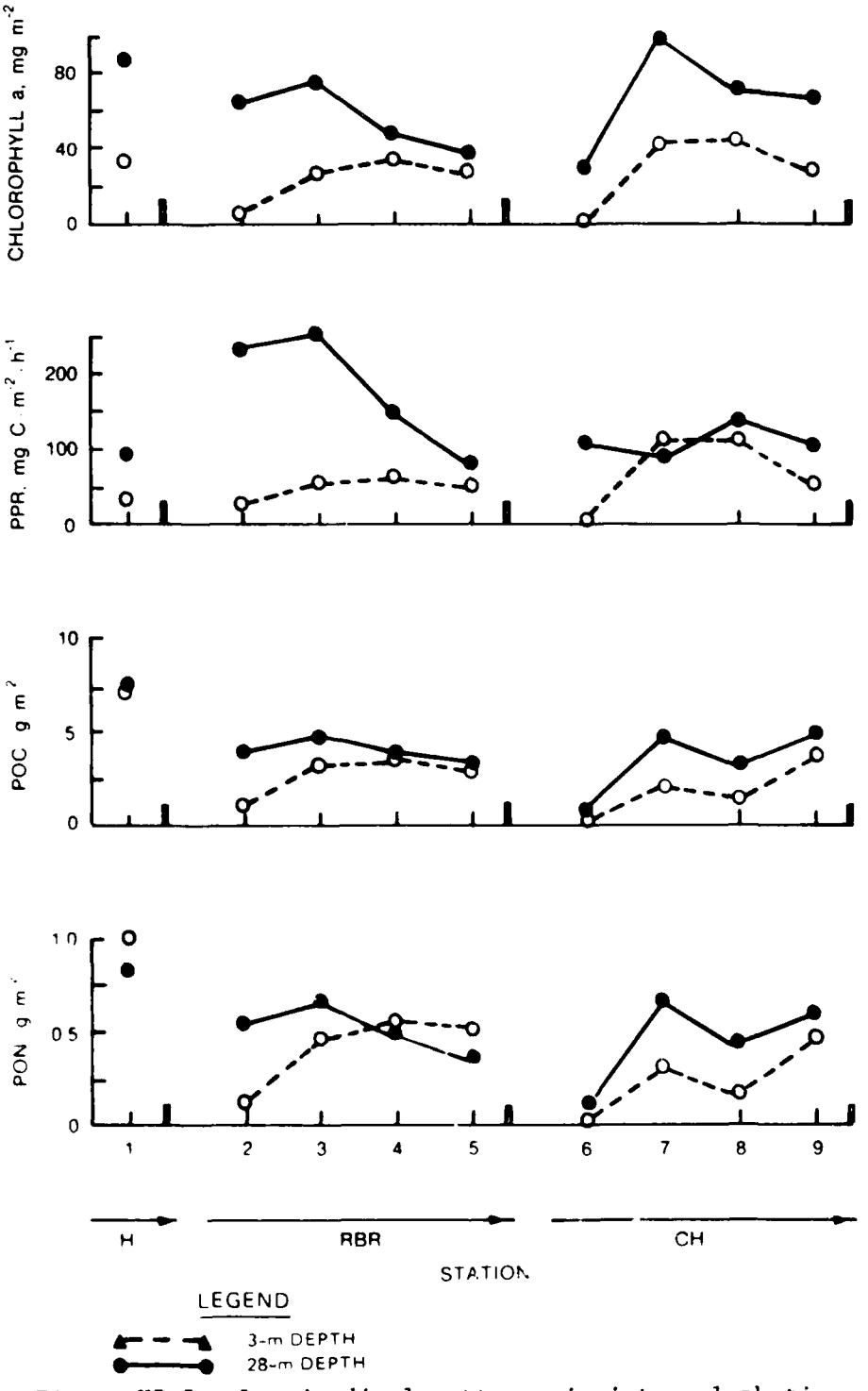

Figure XI-5. Longitudinal patterns in integral photic zone values for chlorophyll a, PPR, POC, and PON in the HT-RBR-CH reservoir series, July and September 1984

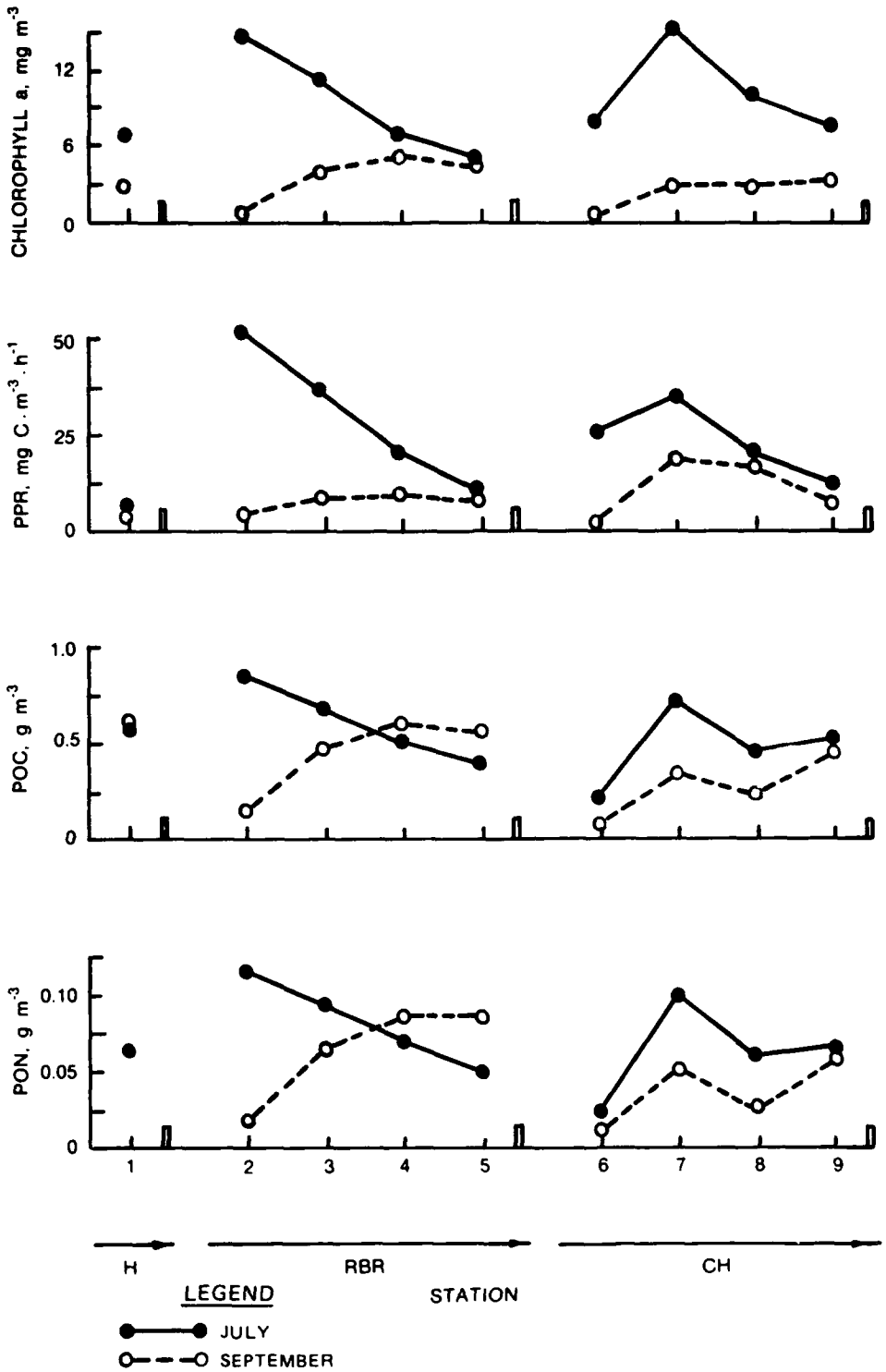

Figure XI-6. Longitudinal patterns in mean photic zone values for chlorophyll a, PPR, POC, and PON in the HT-RBR-CH reservoir series, July and September 1984

272. Because discharge from RBR was from the tainter gates (upper 10 m of the water column) throughout the summer, the RBR water column was very strongly and shallowly stratified. In RBR, the mixed layer depth was only 1 to 2 m in July, as compared to 4 to 5 m in CH and 7 to 8 m in HT (Figure XI-4). In all three reservoirs, the depth of the photic layer (Z_p, the depth to which 1 percent of incident surface radiation penetrates) remained relatively constant between July and September. However, the depth of the mixed layer (Z_m) increased at all stations by September and, therefore, the Z_p:Z_m ratio (i.e., the relative proportion of the mixed layer receiving >1 percent surface light) decreased. However, Z_m increased to a lesser extent in RBR than in HT and CH, and except for station 2 in September, the depth of the photic zone always exceeded that of the mixed-layer depth (i.e., Z_p:Z_m > 1) in RBR. Nevertheless, phytoplankton biomass was limited primarily to the upper 8 m of the water column in RBR during both July and September. In HT and CH, the portion of the water column containing significant quantities of chlorophyll (the mixed layer) deepened downlake to > 20 m near the dam in correspondence to the destabilizing influence of hypolimnetic discharges.

273. In July, Z_p greatly exceeded Z_m in RBR. Because of the penetration of light into the RBR metalimnion and upper hypolimnion in July, the development of pronounced subsurface chlorophyll maxima was anticipated. However, no hypolimnetic chlorophyll peaks were detected in RBR and the metalimnetic peaks in RBR were no more pronounced than those in CH (Figure XI-3). Similarly, because of the extent of anoxia in the RBR hypolimnion, the development of strata of high bacteriochlorophyll concentrations resulting from the growth of photosynthetic green or purple sulfur bacteria was expected. However, hypolimnetic bacteriochlorophyll peaks were not detected in our field fluorometric measurements. Presumably, our continuous method of water column sampling for in vivo chlorophyll fluorescence measurements should have permitted detection of hypolimnetic chlorophyll and bacteriochlorophyll peaks if they were present.

Particulate C:N,
POC:Chl, and PPR:Chl ratios

274. Ratios of particulate C:N and POC:Chl are useful indicators of the relative contribution of detrital materials versus that of algal and bacterial production to the total seston standing crop. Except for a relatively high value (C:N = 10.5) in HT, mean photic zone C:N ratios in the reservoir series did not differ greatly in July (Figure XI-7). Except for the uplake station in CH (station 6), which was influenced by detrital material in the hypolimnetic discharge from HT, seston C:N ratios were relatively low (8.2 to 9.6) and generally increased slightly from uplake to downlake stations in both RBR and CH. In September, uplake stations in both RBR and CH (stations 2 and 6, respectively) reflected the effects of discharges from the dams upstream with elevated seston C:N ratios, which then declined farther downlake. In CH, the C:N ratio declined markedly at station 7 in correspondence with high chlorophyll concentrations, indicating that algal production was a major contributor to the total seston in the midlake portion of CH.

275. The ratio of POC:Chl provides a more direct indicator of the relative contribution of algal biomass to the total seston. For example, low POC:Chl values at metalimnetic depths in CH and RBR (<50 and 50 to 100, respectively) in July (Figure XI-8) indicate regions of relatively high algal production and important contributions of phytoplankton biomass to the total seston standing crop. The POC:Chl ratios increased markedly throughout the reservoir series between the July and September sampling dates (Figures XI-7 and XI-8). Because both POC and chlorophyll concentrations decreased, the general increase in the POC:Chl ratio must have resulted primarily from reduced photoplankton production rather than from an accumulation of detrital particulate carbon in the photic zone. In September, POC:Chl ratios at midlake stations (stations 7 and 8) in CH were <50, reflecting a region of vigorous algal growth and further suggesting that nutrients discharged from the RBR dam stimulated late-summer phytoplankton production in CH.

276. The PPR:Chl ratio indicates the relative physiological status of the reservoir phytoplankton. In general, the PPR:Chl ratio

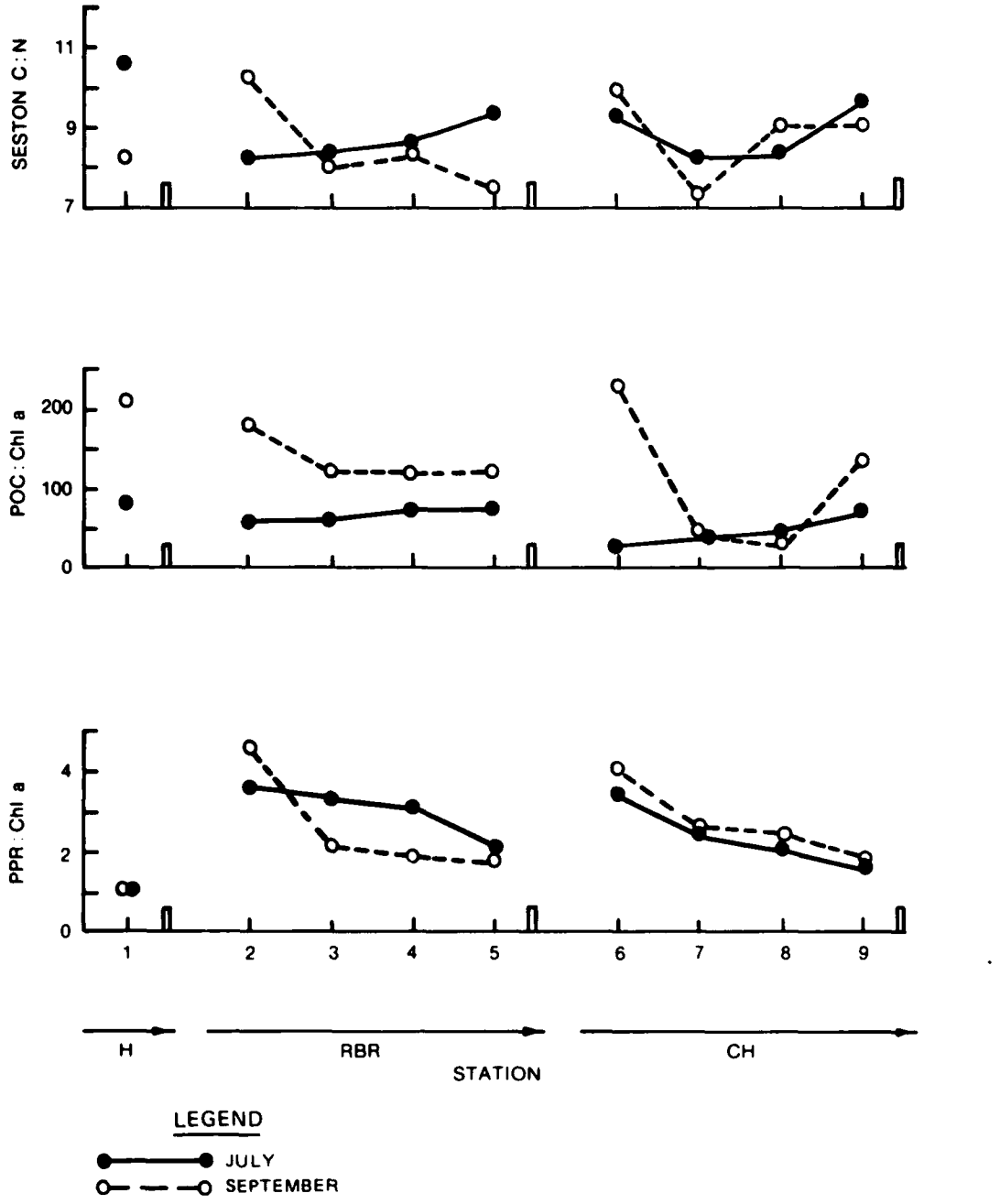

Figure XI-7. Longitudinal patterns in mean photic zone values for seston particulate C:N atomic ratio, POC:chlorophyll a ratio, and PPR:chlorophyll a ratio in the HT-RBR-CH reservoir series, July and September 1984

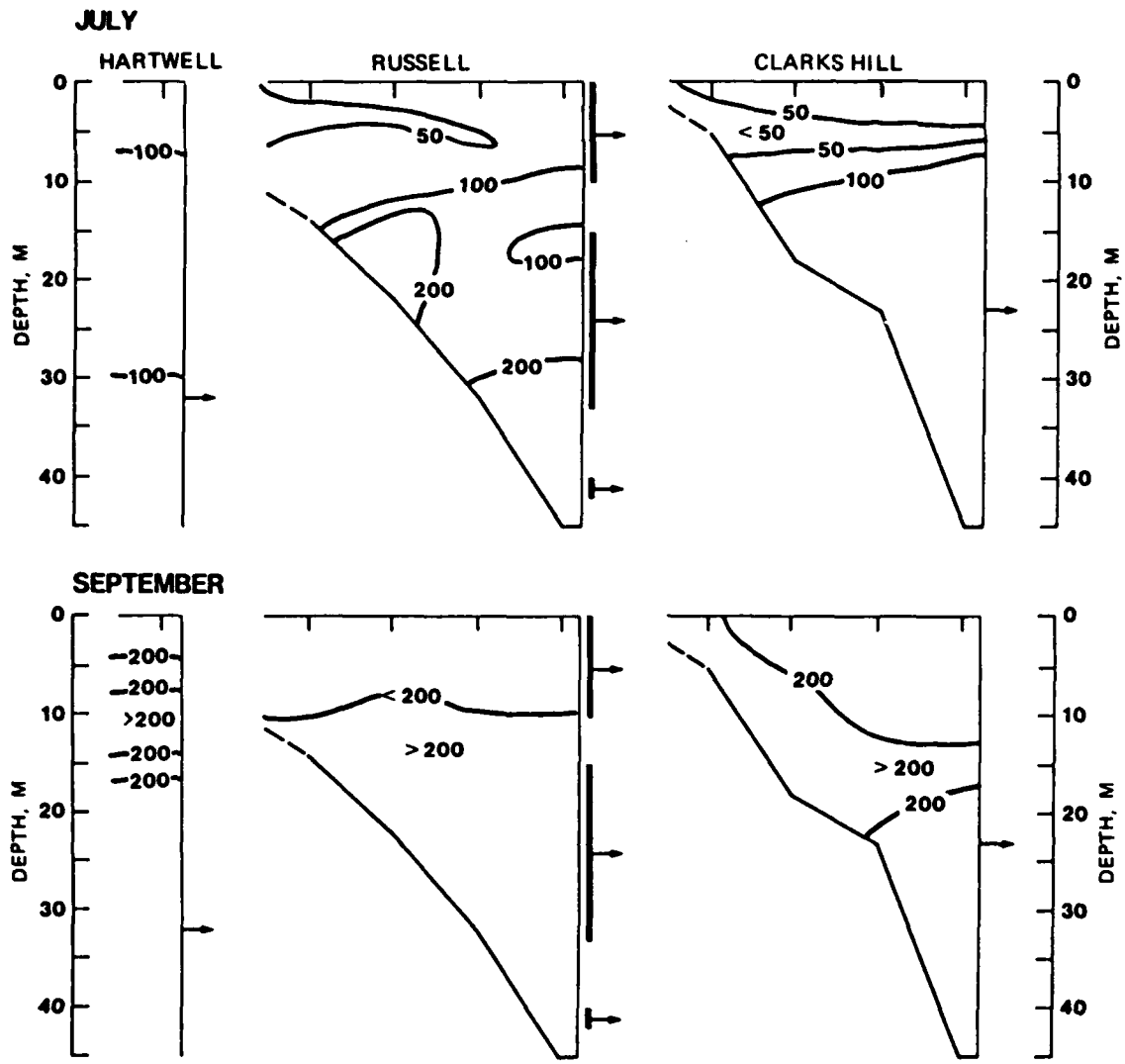

Figure XI-8. Isopleths for seston POC:chlorophyll ratios in the HL-RBR-CH reservoir series, July and September 1984

increased from reservoir to reservoir down the impoundment series, but decreased from uplake to downlake within each reservoir (Figure XI-7). This is the same pattern observed in previous studies of an eastern Tennessee multiple-impoundment series and, we believe, is related primarily to nutrient availability (Elser and Kimmel 1985). It is interesting to note that, except for the RBR uplake station (station 2), which is influenced by the HT discharge, the PPR:Chl ratio decreased from July to September in RBR but was unchanged in HT and CH. It is

hypothesized that early-summer phytoplankton production in RBR was based primarily on nutrients derived from internal loading (nutrient leaching, organic matter decomposition) from the recently inundated basin. By later summer, this nutrient supply had been depleted in the photic layer, and PPR and chlorophyll concentrations in RBR declined to levels more equivalent to those observed in HT and CH.

Dissolved organic carbon
and bacterial productivity

277. Compared to the temporal and spatial fluctuations of seston chemistry and algal biomass, DOC concentrations were relatively invariable in the reservoir series (Figure XI-9). Photic layer DOC concentrations generally increased from reservoir to reservoir down the multiple-impoundment series and from uplake to downlake within individual reservoirs but, unlike algal productivity and biomass levels, DOC did not vary much from July to September. DOC concentrations increased somewhat (from 2.5 to 4.0 mg/ℓ) in the deep hypolimnion of RBR between July and September. However, in view of the large quantity of terrestrial soil, litter, and vegetation recently inundated in the RBR basin, the relative lack of hypolimnetic flushing in the initial year of impoundment, and the extent of hypolimnetic anoxia, it was surprising that DOC concentrations in RBR did not reach significantly higher levels than in HT and CH.

278. One potential explanation for the absence of a marked accumulation of DOC in RBR during the first summer of basin inundation is that the DOC derived from inundated terrestrial organic matter was sufficiently labile to be rapidly metabolized by heterotrophic bacteria. Certainly, the extent of hypolimnetic oxygen depletion observed in RBR during the summer of 1984 relative to that in HT and CH is clear evidence that organic matter decomposition in the new reservoir was very significantly enhanced by the presence of terrestrial organic matter. If this hypothesis is correct, one might expect the standing crop of bacterioplankton in RBR to greatly exceed that in HT and CH as a result of the higher availability of labile organic substrates in the new basin. However, vertical and longitudinal profiles of bacterial cell

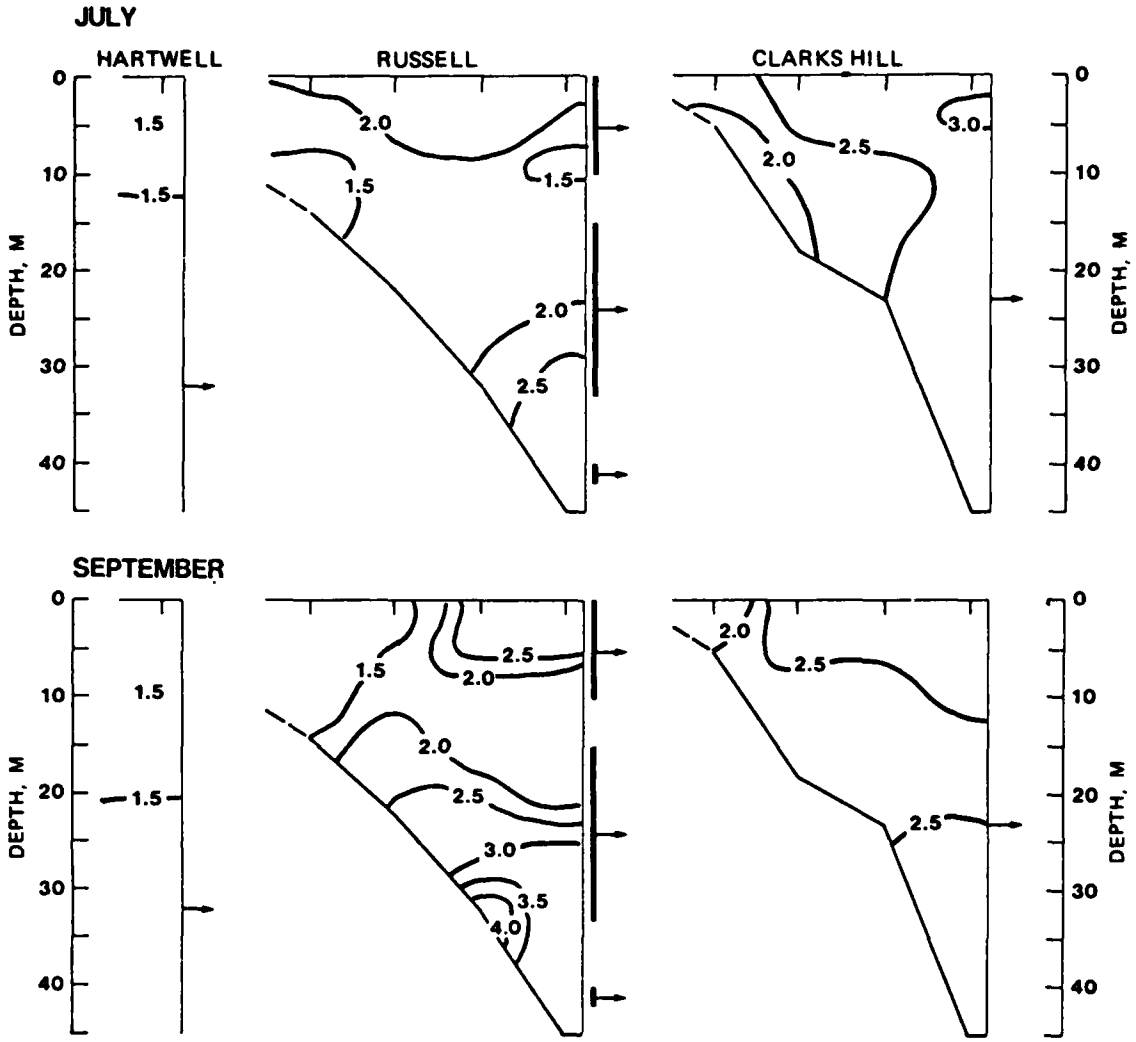

Figure XI-9. Isopleths for DOC in the HL-RBR-CH reservoir series, July and September 1984

numbers show that the bacterial standing crops were equivalent in the three reservoirs (Figure XI-10). The densities of bacteria cells observed in HT, RBR, and CH (1 to 3 million cells per milliliter) are within the normal range for unpolluted natural waters. For comparison, bacterial densities in sewage lagoons are on the order of 1 billion cells per milliliter.

279. Equivalent bacterial standing crops in the three reservoirs do not necessarily indicate that bacterial production is also

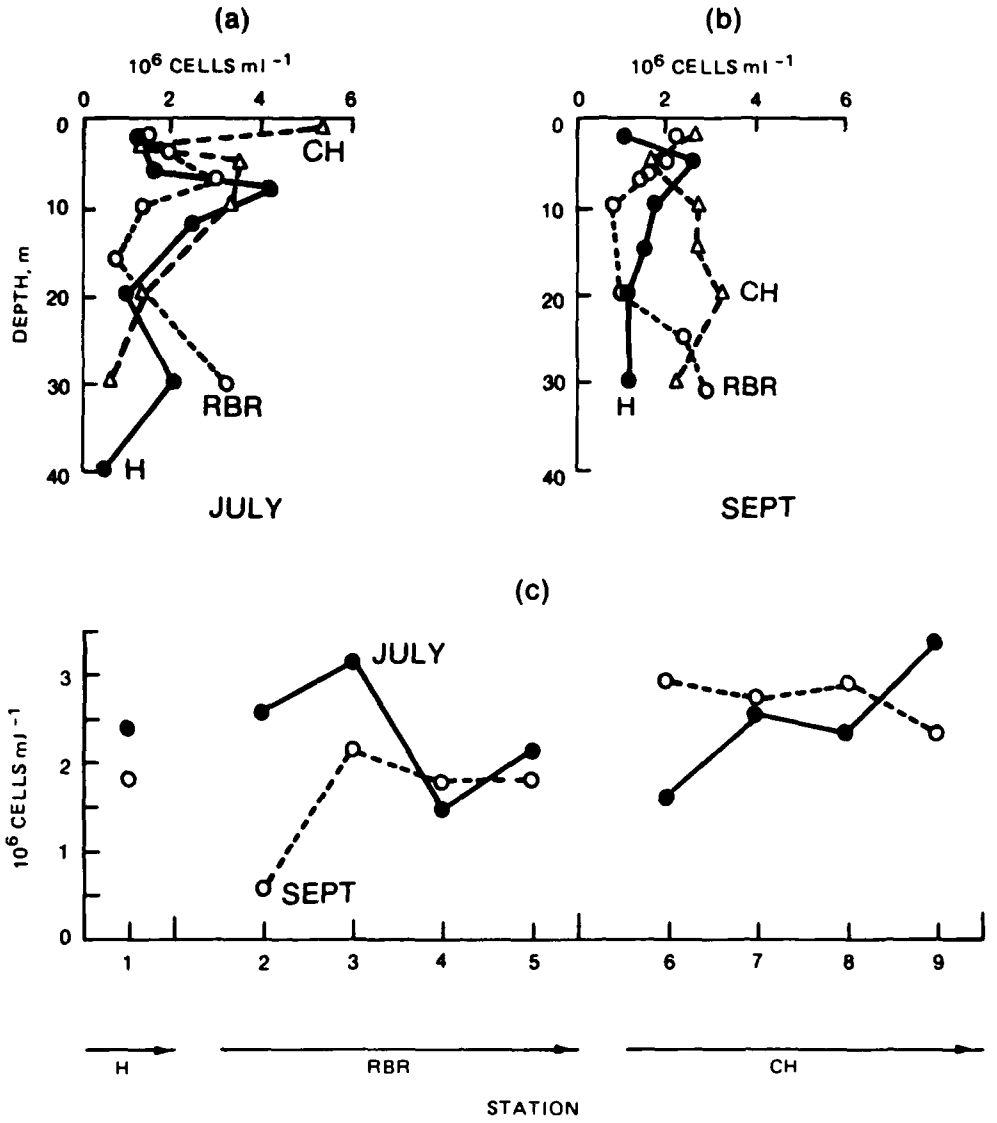

Figure XI-10. Vertical profiles of bacterial cell numbers for downlake station in Hartwell (station 1), Russell (station 5), and Clarks Hill (station 9) Lake during July (a) and September (b) 1984. Mean photic zone densities of bacterioplankton at stations along the longitudinal axis of the HL-RBR-CH reservoir series in July and September 1984

equivalent; i.e., higher growth rates could be offset by higher loss rates. A time-series experiment was conducted in September to directly determine the growth rates of mixed-layer bacterioplankton in the three reservoirs. The results demonstrate that the doubling rate for bacteria

was significantly higher in RBR than in HT and CH in September (Figure XI-11). Because bacterioplankton standing crops were equivalent in

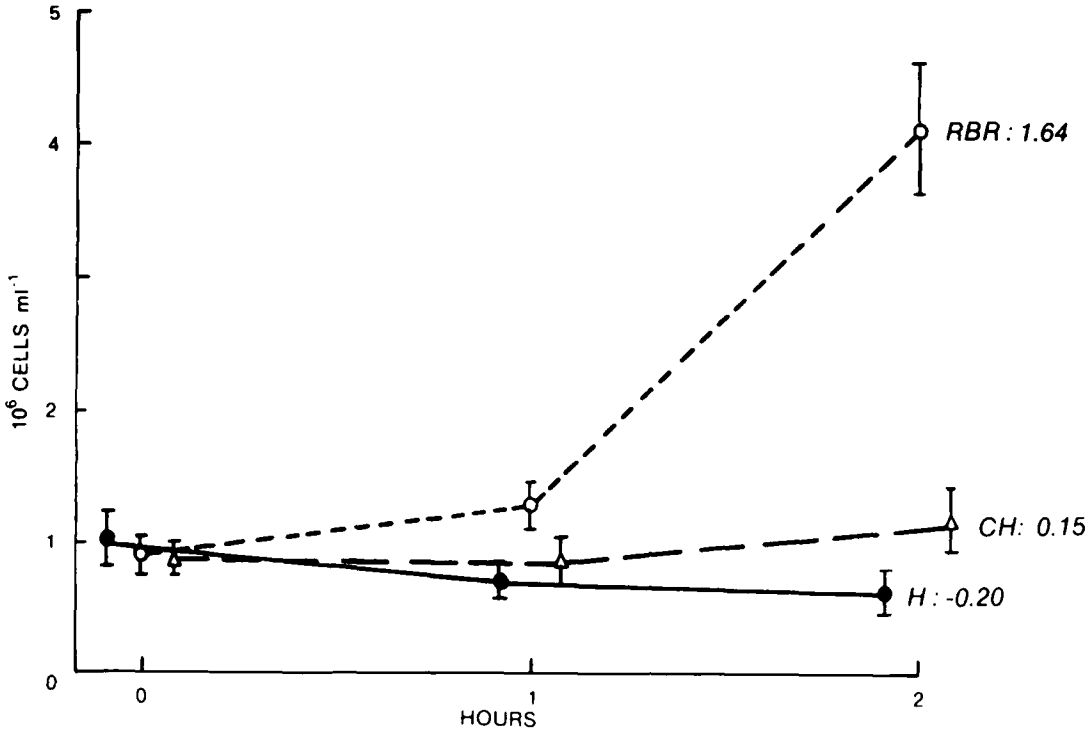

Figure XI-11. Changes in bacterial cell numbers in samples from downlake stations in Hartwell, Russell, and Clarks Hill Lake during time-series incubation experiments on 11-13 September 1984. Estimated bacterial doubling rates (doublings per hour) are indicated at the right

the three reservoirs at the time of the growth rate measurements, it was concluded that the much higher rate of bacterial productivity in RBR must be offset by higher bacterial loss rates, presumably resulting from grazing.

Stable isotope composition of carbon pools

280. The $^{13}C/^{12}C$ data for POC and zooplankton from HT, RBR, and CH support the hypothesis that bacterial production based on the metabolism of inundated terrestrial organic matter was utilized by planktonic grazers and incorporated into the RBR foodweb (Table XI-2). Particularly in July samples, POC in the RBR water column was considerably more ^{13}C-depleted (i.e., had a more negative $\delta\ ^{13}C$ value) than HT or CH

Table XI-2

Stable Carbon Isotope Composition (Expressed as $\delta^{13}C$ Values*) of DIC, DOC, Suspended POC, and Zooplankton from the HT-RBR-CH Reservoir Series, July and September 1984

Date	Reservoir Station and Depth		$\delta^{13}C$ Value DIC	DOC	POC	Zooplankton
7-10	HT-1	6 m	-9.4	-25.4	-22.0	-
7-11	RBR-2	2 m	-7.8	-25.3	-26.4	-
		10 m	-12.4	-21.6	-24.5	-
	RBR-5	2 m	-5.2	-26.3	-24.3	-30.1
		30 m	-9.3	-22.3	-23.9	
7-12	CH-6	0.5 m	-9.4	-21.4	-21.5, -21.3	-
		4 m	-10.1, -10.3	-22.1	-22.9	
	CH-9	1 m	-6.5	-22.1	-22.2	-24.5
		30 m	-11.6, -11.8	-22.4	-22.4	
10-11	HT-1	5 m	-9.6	-25.0	-	-
		30 m	-11.0	-26.1	-	
10-13	RBR-2	2 m	-10.8	-26.6	-29.6	-
	RBR-3	3 m	-7.8, -8.0	-25.5	-28.2	-
		15 m	-9.5	-26.8, -27.0	-27.9	
	RBR-4	2 m	-5.6	-26.1	-27.2	-
		31 m	-4.5	-24.9	-26.0	-
	RBR-5	2 m	-5.1	-27.2	-26.9	-
		31 m	-6.8	-25.6	-25.8	
10-12	CH-6	2 m	-12.8	-28.3	-26.5, -26.5	-
	CH-7	10 m	-9.7	-27.6	-25.9, -25.8	-
	CH-9	5 m	-10.0	-26.0	-25.1	-
		30 m	-10.4	-25.7	-25.6	

* $$\delta^{13}C = \left[\frac{^{13}C/\,^{12}C \text{ sample}}{^{13}C\ \ ^{12}C \text{ standard}}\right] - 1 \times 10^{3};$$

The standard is Peedee belemnite (PDB), a marine carbonate.

samples. Although only two zooplankton samples were analyzed, the $\delta\ ^{13}C$ values for downlake RBR zooplankton were much more negative (-30.1) than downlake CH zooplankton (-24.5) indicating a greater incorporation of terrestrially derived organic matter ($\delta\ ^{13}C$ = about -27) into the RBR planktonic food chain.

281. The DIC from reservoir hypolimnia was usually more ^{13}C-depleted than that from epilimnia (Table XI-2), probably a result of the greater relative contribution of respired CO_2 to the hypolimnetic DIC pool. In July, $\delta\ ^{13}C$ of DOC increased from -25 to -22 downstream from station 1 to station 9 along the longitudinal axis for the reservoir series. This pattern was interpreted to indicate a decreasing contribution of allochthonous organic matter and an increasing contribution of autochthonous organic matter to the DOC pool moving down the reservoir series. This pattern is not as apparent in September as in July, probably due to the accumulation of recalcitrant dissolved organic matter in the mixed layers of all three reservoirs in late summer and to the reduced contribution of algal DOC as PPR decreased in late summer relative to the early summer period of high algal productivity.

282. The $\delta\ ^{13}C$ of POC in RBR and CH samples was less distinctly different in September than in July (Table XI-2). In September, however, POC became gradually less ^{13}C-depleted (from -30 to -25) moving downstream through RBR and CH. These data further suggest a transition from allochthonous to autochthonous sources of organic matter moving down the reservoir series.

Summary and Discussion

283. The major results of the 1984 sampling trips to the Hartwell-Russell-Clark Hill reservoir series are summarized as follows:

a. Longitudinal patterns in seston chemistry, photoplankton biomass, and primary productivity in the HT-RBR-CH series were similar to those observed in an eastern Tennessee multiple-impoundment system (Norris-Melton Hill-Watts Bar reservoirs; Elser and Kimmel 1985). Phytoplankton biomass and productivity levels generally decreased downstream within reservoirs, reflecting

uplake-to-downlake decreases in nutrient availability, but increased downstream from reservoir to reservoir, indicating increased nutrient availability down the reservoir series.

b. The extent of dissolved oxygen depletion observed in the RBR hypolimnion is a clear indication that organic matter dynamics in the new reservoir were significantly influenced by the decomposition of inundated terrestrial organic materials.

c. The lack of a large hypolimnetic accumulation of DOC in RBR, the equivalent standing crops of bacterioplankton in the three reservoirs, the higher rate of bacterial growth in RBR, and the greater ^{13}C depletion in POC and zooplankton in RBR indicate that bacterial production derived from inundated terrigenous organic matter was incorporated into the RBR planktonic food chain.

d. Tainter gate releases from the upper 10 m of the RBR water column minimized effects of the new impoundment on CH water quality during summer and fall 1984. However, downstream transport of nutrients from the upper portion of the RBR hypolimnion stimulated phytoplankton production in the midlake portion of CH during the summer.

284. What about next year? In 1985, peaking power generation will begin at the RBR dam, resulting in a major change in RBR operations from surface releases to hypolimnetic discharges. This change in RBR operations provides a fortuitous "experimental design" for comparing the influences of surface versus hypolimnetic releases on the limnology and ecology of the releasing reservoir (i.e., of RBR), and on the physical, chemical, and biological processes occurring in the receiving reservoir (i.e., in CH) downstream. It is hypothesized that:

a. The extent of hypolimnetic anoxia in RBR will be reduced during 1985 as a combined result of reduced levels of labile terrigenous organic matter in the basin, increased hypolimnetic flushing associated with hypolimnetic discharges from the RBR dam, and operation of the RBR oxygenation system. If levels of labile terrigenous organic matter are reduced in RBR, less distinctive differences in the $\delta\ ^{13}C$ values of POC and zooplankton in 1985 relative to those observed in 1984 are expected.

b. Hypolimnetic releases from RBR will likely reduce the stability of stratification of the RBR water column and, as a result, $Z_p:Z_m$ will decrease in RBR due primarily to

increased Z_m. Although Z_p will decrease somewhat, nutrient availability and phytoplankton production in the RBR photic zone may increase in 1985 over 1984, particularly during the latter part of the growing season.

c. RBR hypolimnetic discharges associated with the initiation of peaking power operations will increase downstream transport of oxygen-depleted water, dissolved nutrients, metals, and organic matter; therefore, the presence of the RBR will have more pronounced water quality and ecological effects on CH in 1985 than in 1984. The degree to which CH water quality is effected will depend largely on the successful operation of the RBR oxygenation system to mediate the influence of decomposition processes on hypolimnetic oxygen concentrations in RBR. The degree to which CH biological productivity is effected will depend on the extent to which nutrients released from the RBR hypolimnion enter the CH mixed layer and enhance phytoplankton production or, alternatively, remain in subsur face density flows isolated from the CH trophogenic zone.

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APPENDIX A: MEASUREMENT TECHNIQUES FOR ESTIMATING FOREST BIOMASS IN FISH TREE AREAS

1. From the one set of postclearing, preflooding, 1:12,000 aerial photographs alone, it was not possible to accurately assess the degree of flooding and consequent amounts of submerged biomass in each fish tree area. Therefore, a method for estimating above- and below-surface biomass was devised based on area size, depth, and timber types.

2. For each area, total acres and acres per stand type were measured on aerial photos, and total biomass was calculated. Fish tree area biomass was separated into understory and overstory components. Understory biomass, assumed to be completely submerged, was assigned an arbitrary value of 12 tons per acre, an accepted median value over a range of southern forest types.* This average value was used because field measurements were not possible due to prior flooding of areas.

3. Amounts of total overstory biomass beneath the lake surface were calculated in the following manner. For each area, the average flooded depth (AFD) of standing trees was determined from topographic information provided on US Corps of Engineers Fish Habitat Area Charts for Russell Lake. Fish tree areas were generally cleared down to the 463-ft contour, a total of 13 ft from full-pool level. This means fish-trees stand in water ranging from 13 ft to some maximum depth, depending on the topography of individual fish-tree areas. For each area, an AFD was computed using the formula AFD = [(462 - Deepest Contour)/2] + 13 . Using this estimate for AFD, the percent of overstory biomass below surface could be estimated. Since a typical overstory timber tree contains 53 percent of its total biomass in the first two logs (32 ft) of the stem,** each AFD represents the proportion of submerged biomass in each area. For example, if AFD was 32 ft in one

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* Phillips, D. R., and J. R. Saucier. 1982. Estimating understory biomass. So. Jo. App. For. 6(1):25-27.

** Phillips, D. R., and D. H. Van Lear. 1984. Biomass removal and nutrient drain as affected by total-tree harvest in southern pines and hardwood stands. J. of For. 82(9):547-550.

area, then the amount of overstory biomass below the lake surface would be 53 percent of the total overstory biomass. Combining understory biomass with the submerged overstory biomass gave the estimate for total submerged biomass in each area.

4. The total of 44,957 greens tons of above-water nonsubmerged fish tree area biomass in Russell Lake represents 7.4 percent of the total forest biomass present in the flooded lake basin. The exact fate of this portion of the total biomass is subject to speculation. Certainly, additions will be made to the submerged biomass total as the above-water tree crowns and stems begin to disintegrate from the action of decay organisms, wind, and waves. However, much of this above-surface biomass will dry enough that it should float, once it falls into the water. Thus, some biomass will be distributed around the lake margins as driftwood and not added to the submerged total.

5. Decomposition begins immediately upon tree mortality. Bark and small branches and limbs will fall first; generally, after 2 to 5 years of inundation, only the larger branches and main stems will remain standing. Rate of decomposition varies depending on tree species. However, most trees in Russell Lake should decay rapidly. Except for highly durable eastern redcedar and the somewhat durable white oaks, the large majority of bottomland and upland hardwoods and pine found in the fish tree areas are not decay resistant. It is expected that within 10 to 15 years, most trees in the fish-tree areas will have decayed enough to break off at the waterline. Where they occur, eastern redcedar along with black locust, red mulberry, and catalpa trees may remain standing indefinitely.*

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* US Department of Agriculture. 1974. Wood handbook: Wood as an engineering material. Agricultural Handbook No. 72, Chap. 3, p 16.

APPENDIX B: REAGENTS AND STANDARDS FOR MERCURY ANALYSES

a. Cadmium Chloride - Stannous Chloride Reagent
Dissolve 50 gm $SnCl_2 \cdot 2H_2O$ and 10 gm $CdCl_2 \cdot 2H_2O$ in warm distilled water. Dilute to 100 ml with distilled water.

b. Mercury Working Solution I
Dilute 1 ml reference solution (1000 ppm) to 1 ℓ with distilled water and 3 ml concentrated nitric acid (1 ml = 1 μg).

c. Mercury Working Solution II
Dilute 50 ml Mercury Working Solution I to 1 ℓ with distilled water (1 ml = 50 ng).

d. Nitric acid, concentrated, reagent grade

e. Potassium Permanganate Reagent
Dissolve 50 gm $KMnO_4$ in distilled water. Dilute to 1 ℓ with distilled water.

f. Potassium Persulfate Reagent
Dissolve 15 gm $K_2S_2O_8$ in distilled water. Dilute to 300 ml with distilled water.

g. Scrubbing Solution
Same as Potassium Permanganate Reagent

h. Sodium Chloride - Hydroxlamine Hydrochloride Reagent
Dissolve 120 gm NaCl and 120 gm hydroxlamine hydrochloride in distilled water. Dilute to 1 ℓ with distilled water.

i. Sodium Hydroxide - Sodium Chloride Reagent
Dissolve 300 gm NaOH and 40 gm NaCl in distilled water. Dilute to 1 ℓ with distilled water.

j. Stannous Chloride Reagent
Dissolve 20 gm $SnCl_2 \cdot 2H_2O$ in 40 ml warm concentrated HCl. Dilute to 160 ml with distilled water.

k. Sulfuric acid, concentrated, reagent grade

APPENDIX C: MERCURY CONCENTRATIONS IN THE WATER OF THE SAVANNAH RIVER AND RICHARD B. RUSSELL LAKE

Date	Transect	Station	Depth (ft)	Total Mercury Conc. (µg/ℓ)	Inorganic Mercury Conc. (µg/ℓ)	Percent Organic Mercury
05-31-83*	190	3	Surface	0.415	0.093	78
	180			0.223	0.050	78
	120			0.186	0.105	44
06-29-83*	190			0.300	0.095	68
	180			0.345	0.220	36
	120			0.080	0.035	91
07-29-83*	190			1.120	0.415	63
	180			0.575	0.120	79
	120			0.335	0.050	85
02-09-84	190			0.098	0.062	36
	180			0.041	0.004	99
	120	2		0.000	0.023	--
	120	3		0.023	0.004	84
	120	4		0.155	0.004	98
03-15-84	190	3		0.076	0.019	75
	180	2		0.035	0.000	100
	180	3		0.035		
	180	4		0.035		
	120	2	10	0.056		
		2	81	0.056		
		3	10	0.035		
		3	85	0.035	0.019	47
		4	10	0.167	0.019	89
		4	85	0.194	0.038	80
05-04-84	190	3	Surface	0.120	0.000	100
	180	2	Surface	0.018	0.000	100
	180	3	Surface	0.030	0.000	100
	180	4	Surface	0.030	0.000	100
	120	2	15	0.030	0.010	67

(Continued)

* Data taken from Nicholas, W. D., 1983 (Dec), "Analytical Techniques for Mercury Speciation in Environmental Samples from Richard B. Russell Reservoir," Master's Thesis, Environmental Systems Engineering, Clemson University, Clemson, S. C.

Date	Transect	Station	Depth (ft)	Total Mercury Conc. (µg/ℓ)	Inorganic Mercury Conc. (µg/ℓ)	Percent Organic Mercury
05-04-84 (Continued)	120	2	50	0.369	0.021	94
	120	3	15	0.030	0.010	67
	120	3	50	0.594	0.223	62
	120	4	15	0.052	0.010	81
	120	4	50	0.880	0.128	85
06-14-84	190	3	Surface	0.032	0.000	100
	180	2	5	0.000	0.000	--
		2	20	0.128	0.035	73
		3	5	0.000	0.000	--
		3	35	0.000	0.024	--
		4	5	0.000	0.000	--
		4	25	0.142	0.024	83
	120	2	5	0.032	0.000	100
		2	75	0.115	0.035	70
		3	5	0.032	0.000	100
		3	95	0.225	0.039	83
		4	5	0.032	0.000	100
		4	100	0.302	0.039	87
07-19-84	190	3	5	0.079	0.166	--
		2	Surface	0.049	0.000	100
		2	20	0.185		100
		3	Surface	0.000		--
		3	40	0.000		--
		4	Surface	0.039		100
		4	25	0.099		100
	120	2	5	0.039	0.034	13
		2	50	0.140	0.000	100
		3	5	0.059		
		3	90	0.230		
		4	5	0.034		
		4	90	0.266		
08-30-84	190	3	Surface	0.031		
	190	4	Surface	0.022		
	180	2	5	0.049		

(Continued)

Date	Transect	Station	Depth (ft)	Total Mercury Conc. (µg/ℓ)	Inorganic Mercury Conc. (µg/ℓ)	Percent Organic Mercury
08-30-84 (Continued)	180	2	30	0.004	0.000	100
	180	3	5	0.022	0.000	59
	180	3	30	0.004	0.000	100
	180	4	5	0.013	0.009	100
	180	4	30	0.000	0.000	--
	120	2	5	0.031	0.000	100
		2	40	0.031	0.009	71
		3	5	0.004	0.000	100
		3	50	0.058	0.019	68
		4	5	0.067	0.000	100
		4	50	0.031	0.009	71
10-02-84	190	3	Surface	0.008	0.000	100
	190	4	Surface	0.015		
	180	2	Surface	0.015		
	180	3	Surface	0.028		
	180	4	Surface	0.009		
	120	2	6.5	0.015		
		2	52.5	0.012		
		3	6.5	0.009		
		3	52.5	0.025	0.007	72
		4	6.5	0.008	0.000	100
		4	52.5	0.009	0.000	100

APPENDIX D: MERCURY CONCENTRATIONS IN FISH FROM THE SAVANNAH RIVER AND RICHARD B. RUSSELL LAKE

Date Caught	Location Caught	Weight (g)	Length (cm)	Total Hg (ppb)	Inorganic Hg (ppb)	Percent Organic Hg
		Bass				
Sept. 83*	Gregg Shoals	93	18	<25	3	-
	SC 184 Bridge	368	28	297	6	98
May 84		58	17	135	7	95
		51	15.5	26	4	84
		111	20	234	28	88
		348	30	257	16	94
		94	19.5	469	13	97
		85	19.5	258	44	83
		194	24	300	32	89
		86	19.5	407	5	99
		136	22.5	506	3	99
Aug. 84	Beaver Dam C.	100	18	468	70	83
		82	17.5	532	320	40
		87	18	595	100	83
		78	17.5	457	223	51
		95	18	640	107	83
		70	17	606	148	76
		111	18.5	639	184	71
		127	20	570	103	82
		338	28	851	152	82
	Cold Water C.	198	21	528	52	90
	Cold Water C.	218	24	459	38	92
	Cold Water C.	190	20.5	483	142	70

(Continued)

* Data from Nicholas, W. D., 1983 (Dec), "Analytical Techniques for Mercury Speciation in Environmental Samples from Richard B. Russell Reservoir," Master's Thesis, Environmental Systems Engineering, Clemson University, Clemson, S. C.

Date Caught	Location Caught	Weight (g)	Length (cm)	Total Hg (ppb)	Inorganic Hg (ppb)	Percent Organic Hg
Aug. 84	Cold Water C.	170	22.5	458	5	99
		171	21	454	158	65
		121	19	430	69	84
		94	18	451	100	78
		89	17.5	580	172	70
		89	18.5	515	52	90
		91	17.5	477	115	76
		78	17	609	58	90
		87	18	609	51	92
		84	17	624	137	78
	Rocky River	133	21	587	52	91
		133	20.5	509	177	65
		90	18	526	42	92
		74	17.5	574	65	88
		82	18	544	100	83
		70	17.5	526	53	90
		70	18	552	36	91
		51	16	545	51	91
		53	17	743	101	86
		118	19.5	459	27	94
		91	18.5	391	34	91
		71	17	426	52	88
		82	17.5	631	42	93
			Sunfish			
Sept. 83*	SC 181 Bridge	79	15.5	78	5	94
	SC 181 Bridge	113	16.5	169	3	98
	SC 181 Bridge	132	17	27	<3	--

(Continued)

Date Caught	Location Caught	Weight (g)	Length (cm)	Total Hg (ppb)	Inorganic Hg (ppb)	Percent Organic Hg
Sept. 83*	SC 181 Bridge	133	17	<25	4	--
	SC 181 Bridge	135	17	51	<3	--
	SC 181 Bridge	135	17	156	4	97
	Gregg Shoals	109	15	<25	4	--
	Gregg Shoals	134	16.5	72	4	94
	Gregg Shoals	147	17	<25	5	--
	Gregg Shoals	130	17.5	113	4	96
	Gregg Shoals	193	20	149	3	98
May, 84	SC 72 Bridge	111	18	296	44	85
	SC 72 Bridge	50	13	170	39	77
	SC 72 Bridge	63	13.5	428	208	51
	Rocky River	170	19.5	38	60	--
	Rocky River	215	20.5	179	24	86
	Rocky River	105	17.5	52	60	--
Aug. 84	Cold Water C.	67	15	355	40	79
		87	15.5	191	138	28
		149	18.5	113	68	69
		148	18.5	212	62	80
		78	15	310	63	43
		125	18.5	111	130	24
		89	15	172	157	40
		127	17	260	26	90
		122	17.5	273	16	94
		72	15	481	38	92
	Rocky River	92	17.5	136	43	68
	Rocky River	113	17	43	16	89
	Rocky River	76	15.5	83	43	48
	Beaver Dam C.	85	16	468	64	86
	Beaver Dam C.	98	17.5	236	79	66

(Continued)

Date Caught	Location Caught	Weight (g)	Length (cm)	Total Hg (ppb)	Inorganic Hg (ppb)	Percent Organic Hg
Aug. 84	Beaver Dam C.	61	16	346	22	94
		149	20	288	73	75
		97	16	444	29	94
		102	18	463	13	97
		101	17	648	52	92
		82	16	450	34	88
		71	14	780	50	94
		75	15	661	62	90
		67	14	624	76	88
		51	13	530	66	88
		52	14	364	20	94
		67	14	468	68	85
		93	16	309	28	91
		63	14	569	55	90
		64	15	839	19	98
		58	14	372	5	99
		50	14	935	6	99
		Yellow Perch				
Sept. 83*	Gregg Shoals	50	16	<25	<3	--
	Gregg Shoals	67	15.5	<25	3	--
May, 84	Rocky River	65	18	104	86	17
		94	19	56	18	68
		70	17.5	55	49	11
		67	18	148	85	42
		137	21.5	146	16	90
		212	24	<15	68	--

(Continued)

Date Caught	Location Caught	Weight (g)	Length (cm)	Total Hg (ppb)	Inorganic Hg (ppb)	Percent Organic Hg
May, 84	Rocky River	66	18	96	55	92
		135	22	76	70	8
		132	20.5	91	8	91
		116	20.5	<20	30	--
		98	20	<22	8	62
		103	20.5	46	10	80
		75	19	82	37	54
Aug. 84		95	20	627	15	98
		119	22	524	20	96
		105	21	384	25	94
		99	20	554	32	94
		60	18	282	<1	100
		64	18	588	<1	100
	Cold Water C.	99	20.5	351	22	94
		75	19	459	8	98
		98	20.5	616	20	97
		100	20	401	178	56
		84	19	329	31	90
		71	19	367	39	89
		81	20	252	57	77
			Catfish			
May, 84	72 Bridge	266	29	428	28	67
Aug. 84	Rocky River	199	22	441	18	96
	Rocky River	113	19	384	30	92
	Beaver Dam C.	59	17	473	93	80

APPENDIX E: MERCURY CONCENTRATIONS IN THE SOIL OF RICHARD B. RUSSELL LAKE AND THE SAVANNAH RIVER BED PRIOR TO INUNDATION*

Date	Transect	Station	Total Mercury Concentration, ppb	Inorganic Mercury Concentration ppb	Present Organic Mercury
05-31-83	190	1	33	22	33
	190	2	9	6	33
	190	3	45	18	60
	190	4	72	27	62
	190	5	153	42	72
	180	1	58	31	46
	180	2	43	19	56
	180	3	20	12	40
	180	4	47	46	2
	180	5	90	54	40
	120	1	130	38	71
	120	2	45	33	27
	120	3	28	10	64
	120	4	41	22	46
	120	5	75	44	41
06-29-83	190	1	52	28	46
	190	2	15	6	15
	190	3	66	31	53
	190	4	67	32	52
	190	5	61	30	51
	180	1	46	21	54
	180	2	13	12	8

(Continued)

* Data taken from Nicholas, W. D., 1983 (Dec), "Analytical Techniques for Mercury Speciation in Environmental Samples from Richard B. Russell Reservoir," Master's Thesis, Environmental Systems Engineering, Clemson University, Clemson, S. C.

Date	Transect	Station	Total Mercury Concentration, ppb	Inorganic Mercury Concentration ppb	Present Organic Mercury
06-29-83	180	3	81	72	11
	180	4	86	70	17
	180	5	86	52	40
	120	1	53	26	51
	120	2	18	16	11
	120	3	12	9	11
	120	4	16	9	44
	120	5	63	105	--
07-29-83	190	1	48	24	50
	190	2	18	3	83
	190	3	45	20	56
	190	4	56	22	61
	190	5	70	18	74
	180	1	40	11	72
	180	2	24	10	58
	180	3	33	14	58
	180	4	81	45	44
	180	5	86	97	--
	120	1	28	10	64
	120	2	14	19	--
	120	3	20	5	75
	120	4	14	9	36
	120	5	78	55	29